

Policy Instruments for Environmental and Natural Resource Management

Second Edition



Thomas Sterner and Jessica Coria

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To Greta for a good past
To Gustav, Erik, and Kalle for a sustainable future
To all of you and to Lena for sharing the present

Thomas

To Magnus for your constant support

Jessica

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Preface

Like the first edition of the book, this second edition is intended to be used by people who are interested in the selection and design of policy instruments for the environment: university professors, undergraduate or graduate students, analysts who advise policymakers, and, particularly, people in countries that have not yet made extensive use of market-based policy instruments. Its purpose is to pull together both the theoretical underpinnings and the distinct practical experiences of policymaking that have evolved in the United States, Europe, and other countries of the Organisation for Economic Co-operation and Development (OECD) and also in non-OECD countries, including some formerly planned economies. A wide range of environmental and natural resources issues illustrate points that are ecologically important or good examples of the principles of policy design. This edition consists of an extensive update and modernization to account for new theory and policy experiences. This applies particularly to climate policies already in place and for recent developments in the area, as well as a deeper and broader presentation of important new topics including: experimental economics, natural resource economics (with intertemporal and spatial dynamics), corporate social responsibility, biofuels and payments for ecosystems services.

To be sure, this book is not an encyclopedia of resource and environmental problems, a pure textbook in environmental economics, or a mere description of policies. Many important issues are not covered at all, or at least not in proportion to their importance. Theoretical issues are presented as refreshers for readers who have studied some economics rather than as rigorous training for future environmental economists. If it were possible to write a “cook-book” with “recipes” for environmental policymaking, then it really ought to be done. However, the ecological, technical, social, and economic realities of environmental policymaking are so complex that there are no simple guidelines. Instead, in-depth understanding of both the economics and the environmental science is necessary to successfully design good policies.

Only a small amount of mathematics is presented in this book. Although mathematical proofs can greatly aid some readers, they can just as well frustrate others. The text was written to be intelligible even to readers who skim the formulas, so detailed mathematical explanations are presented as supplemental information.

The topics of this book include the key theoretical issues, worldwide applications, and various “brown” and “green” issues. Our own personal experiences and prior work have necessarily affected the choice of emphases. We have tried to cover as many countries as possible to illustrate various economic systems and income levels; however, cases from Sweden and the United States are overrepresented among the industrialized countries. In this edition our experiences from Latin America and our contacts with the Environment for Development Network of centers (in Africa, China, and Central America) have also provided much new material from developing countries.

Scientific analysis of policy instruments is not new. However, such analysis tends to concentrate on one issue or aspect at a time and tends to be written for specialists. Literature that systematically covers the whole menu of resource and environmental policy issues in different countries is more sparse, and this book is intended to fit in that niche. It is on the other hand a niche that is growing fast, as we have become aware during the process of rewriting this book. We are grateful that the former edition attracted a great deal of attention and was used in Masters’ and Ph.D. courses across the world. We have created a website with further case studies, forum for debate, suggestions for lecturers and students, and other related resources that will complement this book and help keep it updated. The site will also include material that had been omitted from the first edition, as well as new material or links that we or others use in teaching on the design of environmental policymaking. Please visit the Environmental Economics Unit webpage at the University of Gothenburg at www.economics.gu.se/policyinstruments.

As the original author of the first edition, I would like to thank Jessica Coria immensely. She has taken full responsibility for updating this new edition. It has been a great pleasure and privilege to work with her. We are twin souls in the sense of thinking in very similar ways, and I quickly found I could relax and entrust my dearest (academic) baby to her. At the same time, she has brought many new insights and experiences—as well as important modelling skills—to improve, update, and enrich the book. Many sincere thanks also to: Peter Parks, who coauthored Chapter 22; Martin Persson, who provided valuable input on reduced emissions from deforestation and forest degradation (REDD) (Chapter 27); and Markus Wråke for his contributions on the European Union Emission Trading System (Chapter 6).

Thomas Sterner

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During the past decade, Thomas Sterner has been engaged as a teacher in Sweden and in capacity-building efforts worldwide. The material presented in this book has been extensively tested on students, in courses on natural resources and environmental policy instruments at the University of Gothenburg (Gothenburg, Sweden), at teaching workshops at the World Bank (Washington, DC), and at the African Economic Research Consortium elective course on environmental economics (Nairobi, Kenya). The manuscript or parts thereof were presented at several seminars in economics at the University of Gothenburg, the department of political science at the University of Gothenburg, Resources for the Future (RFF), the Harvard Institute for International Development, and the World Bank. We thank all the participants at these events, who are too numerous to mention. Part of the manuscript was also presented at the Institute of Economic Growth (Delhi, India); thanks to Kanchan Chopra, B.N. Goldar, Shubash Gulati, Srikant Gupta, and M.N. Murthy for valuable comments.

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Abbreviations

CAAA	Clean Air Act Amendments
CARB	California Air Resources Board
CDM	Clean Development Mechanism (of the Kyoto Protocol)
CFC	chlorofluorocarbon
CO	carbon monoxide
CO ₂	carbon dioxide
COP	Conference of the Parties (to the UNFCCC)
CPR	common property resource
DDT	(an effective but dangerous pesticide)
EPA	U.S. Environmental Protection Agency
ESMAP	World Bank's Energy Sector Management Assistance Program
EU	European Union
GDP	gross domestic product
GEF	Global Environmental Facility
GHG	greenhouse gases
GPS	global positioning system
GWh	gigawatt-hour
HC	hydrocarbon
IPCC	Intergovernmental Panel on Climate Change
ITQ	individual transferable (fishing) quota
kWh	kilowatt-hour
LEV	low-emission vehicle
MBI	market-based instrument
MEY	maximum economic yield
MSY	maximum sustainable yield
MWh	megawatt-hour
NAFTA	North American Free Trade Agreement
NGO	nongovernmental organization
NO _x	nitrogen oxides
NPSP	nonpoint-source pollution
ODS	ozone-depleting substance
OECD	Organisation for Economic Co-operation and Development
OTC	Ozone Transport Commission
PM	particulate matter
ppm	parts per million

REP	refunded emissions payment
Sida	Swedish International Development Cooperation Agency
SKr	krona (Swedish currency; 10 SKr $\approx$ US\$1.5)
SO _x	sulfur oxides
TAC	total allowable catch
TCE	trichloroethylene
TEP	tradable emissions permit
TRI	Toxics Release Inventory (United States)
TSPs	total suspended particles
UNDP	United Nations Development Programme
UNEP	United Nations Environment Programme
UNFCCC	United Nations Framework Convention on Climate Change
VA	voluntary agreement
VOC	volatile organic compound
WHO	World Health Organization
WTO	World Trade Organization

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Background and Overview

Global population is growing fast—over 80 million people per year—and has doubled since 1960. Most of that growth is in poor countries. India's population has passed 1 billion, and India may become the world's most populous nation within a few decades. Recent projections indicate that the rate of growth is slowing somewhat, but world population is still projected to reach 9 billion within a few decades. This population growth poses considerable challenges for resource and environmental management.

Definitions, Concepts, and Challenges for Policymaking

The links among population, poverty, growth, resources, and environment are complex, and the mechanisms that determine human fertility and mortality (and thereby population dynamics) are an interesting topic of study.¹ The harsh-but-effective Chinese policy has shown the world that policy mechanisms can affect human fertility and mortality, but can population growth be affected by policies that do not infringe so heavily on personal liberties?

Interestingly enough, population growth appears to be decreasing quickly in most countries. The global average number of children per woman has fallen from about 6 in 1950 to 2.8 in the year 2000. In the richer countries, fertility is typically around 2 children per woman, which means that population will stabilize or in fact slowly decline. Income and education are particularly important determinants of fertility, and thus “development” automatically brings some decrease. The speed of transition depends on many cultural and institutional factors that may lock countries into a form of “demographic trap” in which poverty is both cause and effect of fast population growth. Results of studies exploring the links among institutions such as property, marriage, and inheritance law as well as the more subtle cultural determinants of fertility indicate that policies can and do have a large effect on household decisions, such as whether to marry and how many children to have (Dasgupta 1993). This finding indicates that policymakers may be able to successfully affect the fertility issue, but the sociocultural and personal aspects of fertility and mortality make it a difficult area for policy application.

Besides population, other major determinants of human impact on ecosystems are level of consumption and choice of technology. This concept is neatly summarized by the $I = PAT$ equation, whereby impact depends on population, affluence, and technology (Ehrlich and Holdren 1971).

Market Failures

One frustration of many environmentalists is that seemingly simple solutions to serious environmental problems exist but are never implemented. In this book, we write about policy instruments that are designed to ensure implementation. To begin, policymakers must understand why environmental policy is needed. The reasons include market and policy failures that are interlinked with the evolution of property rights.

Market failure is a technical term that roughly refers to conditions under which the free market does not produce optimal welfare. It is thus a “failure” compared with the abstract model economists make of a perfect market economy. Important examples of such failure include external effects (externalities), public goods, common pool resources, poorly defined or defended property rights, noncompetitive markets, and imperfect (or asymmetric) information. Policy failure may appear to be a simpler concept, but a seemingly neutral concept of welfare underlies it. Policies reflect economic interests, and in some cases, there may not be a single policy that is “optimal” for every group in society. One can sometimes distinguish between corrupt policy and bad policy. The corrupt policy is one that claims to be in the interest of the whole country but actually serves the interest of one group (and may actually do that very successfully). A bad policy is one that intends to enhance welfare in a reasonable way but fails due to ineptitude. Property rights are institutions that can be affected by policy, although the process is typically very slow.

Externalities are nonmarket side effects of production or consumption, such as soil erosion caused by unsuitable agricultural practices (particularly on hillside slopes). The silting of dams and the destruction of coral reefs are real costs, but these costs are not borne by the individuals or corporations that cause the damage. Such situations can be seen as consequences of incomplete property rights: if waterways had owners with a right to clean water, then those owners could sue those who caused the soil erosion and thus internalize the effects.

Public goods are products or services that are enjoyed in common, such as defense and air (clean or dirty). The market tends to undersupply these goods because it is hard to exclude those who do not pay. Instead, political processes are needed, such as the election of a government that collects taxes and finances public goods. Common pool resources also have costly exclusion, but the goods produced with these resources are consumed individually (as private goods). Examples include firewood and fodder, and the resources are often managed as common property. Free riding and other mechanisms that lead to the undersupply of public goods may also lead to the overuse of

common pool resources unless institutions are strong enough to limit access by the users. Noncompetitive markets, monopolies, and oligopolies usually result in nonoptimal supply (e.g., too little may be sold at too high a price).

Of all the market failures, asymmetric information is perhaps the most pervasive. Economists typically point out that there are no “free lunches” yet commonly assume that information is freely available to everyone. Information is costly, and lack of information stops the market from operating perfectly. Understanding information asymmetries not only helps us design policy instruments to address monitoring difficulties; it also goes to the heart of the most essential dilemma: how to promote social goals such as equity without destroying incentives for work and efficiency. Because policymakers do not have reliable data on pollution damages and abatement costs, for instance, they cannot design policies that are both efficient (with respect to resource allocation) and fair (in sharing the burdens of all the costs involved). If policymakers need the cooperation of individuals who have “inside” information, then they must accept that those individuals may be able to earn something in return for disclosing information.

Social Rights and Norms Concerning Nature

The concept of environmental problems sounds simple enough, and depending on one’s background, it may bring to mind issues such as factory smoke, soil erosion, and dam siltation. However, at a deeper level, the concept is difficult to comprehend because it touches on the relationship not only between human beings but also between humans and nature.

To determine what an environmental problem is and what needs to be remedied, policymakers must understand not only technology and ecology but also the sociology, economics, and politics of property rights. Rights, policy instruments, and politics are interlinked in ways that vary between economies, and information also plays a pervasive role. One everyday illustration of rights is cigarette smoking.

A few decades ago, individuals had the right to smoke almost wherever they pleased. People who suffered from the effects of secondhand smoke had no alternative but to try to avoid smokers. Over time, increased information and other factors have changed this situation so much that today, in some countries, the rights have been reversed: individuals have the right to enjoy a smoke-free environment. This sea change has permeated even the private sphere, so smokers visiting private homes kindly ask permission to smoke, or they simply go outside before lighting up. The use of instruments such as no-smoking zones, tobacco taxes, prohibition of tobacco advertising, and legal suits against the tobacco companies has strongly affected the general perception of rights regarding cigarette smoking. Whereas some policy instruments are only possible thanks to changes in individual rights, instruments also can help to change the structure of rights by changing moral and ethical perceptions.

Current Problems and Warning Signals

A few examples illustrate the kinds of problems that face humanity:

- Over the course of this century, net carbon uptake by terrestrial ecosystems is likely to peak, thus amplifying climate change.
- Earth's protective stratospheric ozone layer has been degraded by the emission of toxic synthetic chemicals into the air.
- Synthetic chemicals and toxic metals have spread to the supposedly most inaccessible corners of the planet, including the Antarctic; some have accumulated in the food chain and have penetrated the genetic makeup of the human population.
- Already in the 1980s, human activities used about 40% of the primary energy transformation through photosynthesis, which is the basis of all life on Earth. This consumption level does not leave much for natural ecosystems and biodiversity (Vitousek et al. 1986, 1997). Energy consumption, especially of fossil fuels, poses threats at local and global levels. Its potential effects on climate are a topic of international concern.
- Water scarcity is a threat to agriculture and consumers in many countries. The level of some of the world's major waterways (e.g., Nile, Indus, Ganges, Colorado, and Yellow Rivers and the Aral Sea) has fallen visibly as a result of industrial, agricultural, and residential use, and water tables in many regions of the United States, India, China, and other countries are being drawn down rapidly.
- Soil degradation, loss of forest cover, and threats to the marine and coastal ecosystems (e.g., mangroves and coral reefs) have created considerable risk to biodiversity as well as to the sustainability of the food chain.
- Yields of many of the world's fisheries are decreasing. To keep up catches, earnings, and employment, fishermen have stepped up efforts by using larger boats, nets with smaller mesh, and sophisticated technologies such as sonar and satellite navigation. Instead of encouraging restraint, many policies "help" the fishermen by subsidizing the purchase of boats and technology, thus lowering costs to fishermen and increasing the overall fishing effort—thus exacerbating the problem rather than resolving it.
- The energy crisis of the 1970s spurred research into technologies for saving energy (e.g., fluorescent lighting, heat pumps, "hypercars," and thyristors) and for alternative methods of producing energy (e.g., wind power, solar power, and biofuels); good technologies have been developed for efficient energy use in transportation, lighting, heating, and industrial processes. However, sometimes the consumer price of energy is too low to make the alternative technology commercially viable. External costs related to local and global environmental problems (e.g., health and productivity costs of getting asthma and bronchitis in urban areas) usually are not included as part of the cost of electricity or gasoline. If consumers were required to

pay the real total cost of energy, they would be more motivated to adopt energy-efficient techniques.

- People whose livelihood depends on natural resources (e.g., grazing lands) typically know their resources well and would have the knowledge to manage those resources rationally, even optimally, if given the opportunity and the means. However, absolute poverty makes the risk of variations in yield unacceptable and can result in unsustainable behavior. Instead of investing in new productive and sustainable technology, for example, poor individuals might continue to use methods that damage the ecosystem. These methods may be individually rational adaptations that fill the place of missing markets or institutions for savings and insurance, thus showing the detrimental effect of this market failure.
- The income and equity aspects of environmental issues and policy instrument design are often crucial. Imposing taxes to reduce herd size, overfishing, or vehicular traffic can solve congestion and overuse problems but may still be resisted because they leave the users with less welfare if the taxes collected are siphoned off for purposes that are perceived as unproductive for the local users. Policy instruments must give local users a price signal that internalizes externalities without transferring the money out of the local community. There are numerous ways of doing this—for example, through permits that are allocated freely to local users, or by levying charges rather than taxes and then using the charges for local environmental or resource funds, which then can be allocated locally. Many environmental fees in developing countries operate in this way (see Chapter 21).
- In many instances where environmental policy is warranted, polluters have more information and typically greater resources at their disposal than the policymakers do; informational instruments may be an important first step toward successful policy. By collecting and disseminating information, an agency can create a baseline for future action; encourage transparency in implementation, so that individual inspectors cannot “make deals” with polluters outside the law; and clear the way to inform and empower customers, workers, investors, neighbors, and other concerned groups (see Chapter 20).

Applying Theory to Nature

Environmental economics (or ecological economics²) addresses the interface between economics and the life support system of Earth. Natural resources economics addresses both geological resources such as oil and minerals and, increasingly, biological resources such as forests and fisheries. It can be considered an integral part of environmental economics, even though it often is treated as a separate discipline. To take advantage of the lessons that these two areas can provide for each other, we discuss them jointly as far as possible.

Environmental policy is interdisciplinary; although economic theory can make a fundamental contribution to the understanding of policy instruments, it can do so only in conjunction with natural science, technology, and other social sciences.

Some people doubt that the conventional paradigm is forceful enough to manage the many serious environmental problems that now face global society, but an increasing number of powerful policy instruments are now available in the conventional tool kit: taxes, charges, permits, deposit–refund systems, labeling schemes, and other information provision systems. The main problem is that these instruments are rarely used properly. Historical examples of serious attempts at environmental policymaking are quite rare. Rather than worry about whether the available policy instruments will ultimately be sufficient, policymakers should make larger scale use of them.

In attempts to avert ecological disasters, policymakers must remember that “disaster” is already an apt description of everyday life for many people in developing countries. Many of the problems that low-income people face are deeply intertwined with the degradation of natural resources and, in some cases, the spread of pollution. Policymakers must focus on the interaction between poverty and ecosystem resources and take particular care to study the distributional characteristics of environmental and resource issues, especially of proposed policy instruments.

Many developing countries lack the resources needed to implement ideal market-based instruments; for the same reason, they also lack the ability to manage other policy instruments. (Regulations need monitoring, enforcement, and occasionally sanctions, which are not necessarily easier to implement than taxes.) At the same time, the welfare effects of environmental degradation can be the worst and the urgency of economic efficiency the greatest in developing countries. The selection and design of policy instruments is more complicated and more important in developing than in developed countries.

A body of scientific analysis on policy instruments already exists. The many references in this book are only a sample of the available literature. Much of this work concentrates on one issue or aspect at a time and tends to be written for specialists; the seminal article by Weitzman (1974) is a fine example. Other work includes the popular textbook by Baumol and Oates (1988). Tietenberg wrote a series of empirical and conceptual analyses (e.g., Tietenberg 1990), and Xepapadeas (1997) wrote a theoretical book of great clarity. Other central works on instrument selection include that by Bohm and Russell (1997) (see also Supplemental Reading at the end of this chapter).

There is also a specialized literature on environmental policymaking in developing countries. Several important contributions stem from the Harvard Institute for International Development (e.g., Panayotou 1998; Vincent et al. 1997). The World Bank (2000) provides an exciting summary and discussion of many new policy initiatives that its research department has been following and, in some cases, fostering. The authors of such books commonly are either

proponents or strong skeptics of “economic” policy instruments in developing countries. To some extent, this debate may center on whether the glass is half-full or half-empty. However, one should not be too quick to reach general conclusions about which type of instrument is best suited. Choices should be made carefully, on a case-by-case basis.

Overview of the Book

Parts I through V are devoted to the analysis of environmental economic issues, defining the need for policy instruments, reviewing the policy instruments available, and discussing the selection of policy instruments under various conditions. Part VI focuses on the management of natural resources, illustrating the theoretical concepts by looking at instrument choice and design for water, waste, fisheries, agriculture, forests and ecosystems.

Part I: The Need for Environmental and Natural Resource Policy

Chapter 2 presents the classical issues of growth, welfare reform, market failure, externalities, public goods, congestion, asymmetric information and uncertainty. Some of this material is traditional public economics and may be familiar to economists, who may want to skim this part as an introduction. An understanding of public economics is essential to seeing environmental policymaking as one kind of public policy reform. Chapter 3 treats the evolution of rights, which is fundamental to the functioning of markets, the existence of market failures, and the design of market reform.

Part II: Review of Policy Instruments

The main role of Part II is to illustrate the range of available policy instruments and how they operate. Its starting point is the policy matrix that organizes information about various policy instruments and their applications in different areas. Direct regulation is presented in Chapter 4. Other instruments discussed include taxes (Chapter 5), permits (Chapter 6), and subsidies and other instruments (Chapter 7). Details in instrument design are important, so we differentiate kinds of permits depending on how they are allocated. Charges that are refunded to the polluters are treated separately, because they result in a different distribution of the cost burden, and thus the politics of implementation changes. Chapter 8 interprets the notion of “instrument” in a broad sense, including common property resource management and the creation of property rights in general.

Part III: Selection of Policy Instruments

Part III concerns the selection and design of instruments. Chapter 9 focuses on the efficiency of policy instruments under different conditions concerning abatement and cost curves, the character of technical progress, and so on. Chapter 10 examines the role of uncertainty and information asymmetry. The next few chapters present economy-wide (general equilibrium) effects (Chapter 11), effects that are related to income distribution (Chapter 12), and effects of property rights, politics, culture, and psychology on instrument selection (Chapter 13). Chapter 14 is a discussion of international aspects, transboundary pollution problems, and climate change, and Chapter 15 synthesizes the information presented in Part III for application to policy design.

Part IV: Policy Instruments for Road Transportation

Part IV concerns the road transportation sector. The environmental damage caused by transportation and environmentally differentiated road pricing as the corresponding “first-best” policy instrument are presented in Chapter 16. In Chapters 17, 18, and 19 we describe “second-best” policy instruments, which range from regulations and fuel taxes to fuel quality, including also vehicle inspection and maintenance programs, and urban planning in developing-country cities. Chapter 19 also presents a summary of lessons learned from policy experience in road transportation.

Part V: Policy Instruments for Industrial Pollution

The focus of Part V is the design of policy instruments for industrial pollution. Chapter 20 recounts the experience of developed countries (mainly those in the Organisation for Economic Co-operation and Development [OECD]), comparing taxes and permits for acidifying emissions and comparing regulation, prohibition, and taxation. Global issues related to CFCs (chlorofluorocarbons) and climate change are also discussed. Chapter 21 focuses on the experience of developing and transitional countries. Taxes and differentiated tariffs are important, but the focus is on the use of the tax proceeds and on distributional effects. Tradable permits, voluntary agreements, and information provision are prominent instruments, whereas monitoring, funding, and the building of institutional capacity in the environmental protection agencies are fundamental concerns or constraints for policymakers.

Part VI: Policy Instruments for the Management of Natural Resources and Ecosystems

The overarching theme of Part VI is the management of natural resources and ecosystems. Chapter 22 comprises the theoretical background on the intertemporal management of natural resources, spatial and ecological complexities

that are sometimes underestimated in applying economic models to environmental policymaking. Chapter 23 focuses on water, Chapter 24 on waste, Chapter 25 on fisheries, Chapter 26 on agriculture, Chapter 27 on forests, and ecosystem services are examined in Chapter 28. These issues are of the greatest significance to people and countries with low incomes, because natural resources can be the main source of livelihood and future prospects for people in countries that have little industry. However, the underlying technology and science is complex and often poorly understood, and many categories of users have fairly insecure or unclear rights—highlighting the importance for welfare of distributional issues. In addition, the political and cultural setting can be complex and conflict-ridden.

The text ends with Chapter 29, which attempts to summarize some of the main issues of environmental policymaking and their potential solutions.

Additional Materials

Even though abbreviations are defined in the text, a list of common and technical abbreviations appears toward the front of the book. Because readers of this book come from different academic backgrounds, Supplemental Reading lists are provided at the end of most chapters. The comprehensive References list at the end of the book includes full citations of the bibliographical references cited in the text as well as the Supplemental Reading listings.

Supplemental Reading

Environmental Economics and Policy

Dasgupta and Mäler 2000
Freeman 1993
Hanley, Shogren, and White 2007
Kolstad 2010a

Relationship between Population and Resources

Bartza and Kelly 2008
Dasgupta 2000

Gallager 2009

Jodha 1988, 1998
Luck 2007

Selection of Policy Instruments

Dijkstra 1999
Russell and Powell 1996
Stavins 2001
Sternier 1994
U.S. EPA 2001

Notes

- 1 The most important link is perhaps between the environment and the number of rich people, who consume more and thus exert more pressure on the ecosystem. This effect is sometimes referred to as an ecological footprint. It has been estimated that if 6 billion people were to enjoy a North American standard of life,

then the equivalent of another two planets would be needed to meet the demand for resources (Rees and Wackernagel 1994). This eye-opening observation can be misleading because it assumes constant technology, whereas the ecological impact of economic activity depends crucially on technical progress. In other words, many extra planets would already be needed to meet current consumption if the technologies prevalent 50 years ago were still in use.

- 2 Ecologists and natural scientists tend to call themselves “ecological” economists, whereas economists appear to prefer the term “environmental.” For some researchers, there is an ideological difference between the terms, which are not identical but do overlap strongly; the distinction is not emphasized here.

PART I

The Need for Environmental and Natural Resource Policy

The first part of this book is dedicated to explaining why environmental and natural resources problems exist. This task is not as trivial as it may sound, because many potential answers can be given. In fact, a term like “environmental problem” is really a misnomer. It is not the environment that creates problems for society but society that creates problems for itself by not understanding how to interact with the environment. To address these issues, a chemist or physicist might concentrate on the spread of certain compounds, whereas an ecotoxicologist might analyze the resilience of an ecosystem to certain disturbances. A social scientist might study laws and norms, and an architect might be concerned with town planning. Each of these viewpoints contributes a vital element to our collective understanding.

In the next four chapters, we concentrate on conveying how economists address environmental and natural resources problems, starting in Chapter 2 with the most important concept: market failure. Economists generally believe that markets can be very efficient at allocating resources, but under many conditions, their attractive efficiency actually breaks down. In the area of the environment, this scenario is unfortunately common; therefore, it is important to consider very carefully what role markets can play and how their efficiency in allocation and their fairness in distribution can be enhanced by policymaking. Chapter 2 also reviews the economics of public

policy and information, and Chapter 3 discusses the evolution of property rights to ecosystem resources.

Consequences of Economic Growth

The debate on the consequences of economic growth for humanity dates back at least to Thomas Malthus (1766–1834), who prophesied that population would grow exponentially and resources would be constant or grow more slowly (linearly), so that people would be doomed to live in poverty (Malthus 1803). More recently, the Club of Rome researchers (often referred to as “modern Malthusians”) have warned that even the current population and economic activity on Earth is unsustainable.

The first Club of Rome report discussed shortages for such vital metals as lead and mercury that have not materialized (Meadows et al. 1972). Today, there is widespread concern about the toxicity of these chemicals and their abundance in the ecosystems, not their scarcity. This example illustrates an important economic mechanism: with market prices, tendencies toward a shortage—or increased demand that could lead to a shortage—tend to raise the price, which leads to substitution away from that particular commodity and increased resources spent on discovery and developing alternative supplies. To date, this mechanism appears to have been sufficient to avoid shortages of lead and mercury. However, the mere existence of this mechanism does not negate the importance of a country’s carefully considering how to manage its natural resources to give maximum benefit, which includes formulating contracts with contractors, designing regulations, and determining resource rents.

The fact that the Club of Rome researchers were wrong about the supply of certain minerals does not mean that they will be wrong about ecosystems in general. In the realm of renewable resources (e.g., fisheries, forestry, water, and agriculture), overuse is a real possibility because of the combination of complicated ecology and inappropriate property management systems. Poor countries must develop policies that both bring about economic growth and address environmental concerns (World Commission on Environment and Development 1987). Global environmental threats include climate change, ozone layer depletion, acidification (of lakes, forests, and more), and the spread of both synthetic (human-made) chemicals and toxic elements from the geosphere into the biosphere. Local environmental threats, which often have health effects, include noise, air pollution, unsanitary working conditions, and infectious disease related to poor water and waste management.

Additional environmental issues include natural resource degradation, overfishing, the destruction of forests, damage to marine ecosystems, soil degradation, and overgrazing of commons. Although local in focus, these issues create a global problem in that the resources are essential to the livelihoods of billions of people in communities around the world. These issues also are inextricably related to each other and to other environmental problems

in numerous ways: for instance, soil erosion can exacerbate air pollution and water pollution, which affects fishing, recreation, and human health. Intensive agriculture requires pesticides, which may cause ecological damage and human health problems while leading to decreased overall biodiversity (yet another important global issue). Deforestation dramatically increases the damage from hurricanes and other natural disasters.

The above list is by no means exhaustive. Unfortunately, many other environmental and resource problems exist—many of which are still poorly understood, and some of which have not yet been identified.

Institutional and Policy Failure

Much of Part I is dedicated to the discussion of market failure. However, market failures are not always the most serious threats to ecosystems. Institutions are also imperfect, and one of the most important institutions is government. The monumental failure of state ownership has already been mentioned. The notion that the state is a neutral and perfect agency to enforce the general well-being of society is very naïve.

One example of imperfect government policy is the formerly planned economies of eastern Europe, where the banishment of “short-sighted profit interests” was hailed as an opportunity to implement policies truly geared to maximizing welfare; yet, the policies really achieved the exact opposite goal, partly because of a simplistic application of the Marxist theory that value is created only by labor. By treating natural resources as free goods of no value, the intrinsic value of those resources was, in many cases, effectively destroyed. The Aral Sea is a sad symbol of such policy. More than half of its area is gone. This giant lake has been turned into a poisonous dust bowl as a result of irrigation, poor management, and excessive cotton production. Ships now lie in the sand, many miles from the current shoreline.

Another example of imperfect government policy is fisheries, because the absence of property rights to the sea leads to the risk of a tragedy of open access. It is a case of considerable market failure in which political policies are badly needed, but the wrong policies make matters worse. Subsidies intended to “help” fishermen actually exacerbate the market failure that they are supposed to address.

In the future, the public sector must not be analyzed as if it were a monolith. In reality, government consists of a series of public-sector bodies with distinct structures, motivations, and modes of operation at different levels. Furthermore, governments are not the only institutions that can fail. The family is also an institution at the micro level, and some form of rational division of labor within it (as well as some fair division of proceeds among its members) is often assumed. However, it is not always the case. In many poor communities, men hold a large share of formal rights and power while women and children are effectively dispossessed and exploited within the contexts of

their own families. It is not uncommon for the government to exacerbate such tendencies by automatically granting titles to a male “head of household” when traditional rights may have been much more equal, even where agriculture was managed primarily by women.

Some faulty government policies are a real threat to the sustainable use of natural resources and ecosystems—for example, subsidies for goods, services, or practices that cause severe environmental degradation. Such subsidies are so common that many economists (particularly those working with developing countries) rank “subsidy removal” as one of the main instruments of environmental policy. Given this sadly ironic state of affairs, one might expect that this section ought to be one of the longest in this book. However, undoing bad policies is not a new category of policy; rather, it is part of a general process of policy optimization or adjustment.

One possible, and indeed plausible, explanation for the prevalence of bad policy is lack of information or understanding about the ecological, technical, and economic relationships that are used to choose and design policy instruments. Other explanations stem from the fact that policies are not designed only by altruistic welfare-maximizing policymakers who are free from personal economic or political interests. The truth is, policies are formed by the interplay of conflicting political and economic interests, and a thorough understanding of the political economy of policymaking is required to analyze any set of instruments used in a specific context. In countries where state institutions are weak, the risk of policy capture by various groups is particularly strong. Presumably, the abundance of damaging subsidies for agriculture and industry in developing and developed countries should be seen in this context.

Causes of Environmental Degradation

How are environmental threats related to economic development and growth?¹ At one extreme, some researchers claim that growth has already exceeded the sustainable level of activity on Earth (Meadows et al. 1972). At the other extreme, some researchers believe that technical progress will make it possible to meet the demand that will result from increased population and per capita income (Kahn et al. 1976). In some overly simplified analyses, growth is the main culprit, whereas in others, it is the principal panacea for environmental issues. Neither of these simple positions is tenable: the truth is that we need growth not least for poverty alleviation but must avoid the kind of growth that leads to ecological damage that in turn will threaten welfare. We need to focus not on the simple yes or no to growth but on the sectoral and technical characteristics that will give us sustainable growth. In this chapter we focus on the links between economic growth and the environment and the circumstances when the market mechanism is inefficient at allocating environmental resources.

Growth and the Environment

The essential determinant of environmental stress is not the average rate of growth but the technology used and the composition of growth or of the economy itself. Whereas increased consumption of polluting cars, pesticides, and chemical-intensive products could become a problem for sustainability, increased consumption of music, Internet information, ecotourism, and organically grown food probably would not. However, in a free market economy, the consumers—not the “social planners” or the ecologists—decide the composition of output (and, indirectly, production).

Policymakers can influence the path of the economy by using policy instruments. To do this, they must first understand the fundamental determinants of economic development. A great deal of research has focused on the composition of the economy and its development over time, which is often associated with the environmental Kuznets curves (EKC). The idea behind EKCs is

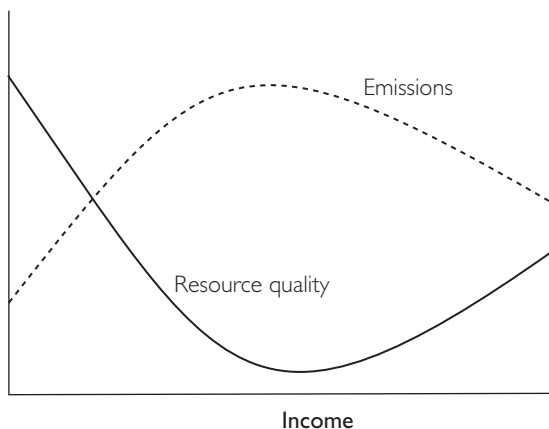


FIGURE 2-1 An Environmental Kuznets Curve

that with economic growth, emissions typically follow the inverted “U” curve (as illustrated for emissions in Figure 2-1). According to this hypothesis, the early phases of economic growth inevitably imply increased pollution, but as incomes increase, emissions peak and then decline. The curve for the quality of ecosystem resources would be the inverse of that for emissions—that is, an upright “U”—signifying deterioration followed by gradual improvement. However, the stock character of these resources complicates the issue.

Explanations for the EKC concentrate on several general factors, including the development of technology, relative prices, the income elasticity (which describes how demand for goods and services tends to vary with income) for a clean environment, and income distribution. Income distribution also varies with economic growth, as illustrated in the original Kuznets curves (Kuznets 1930), adding another layer of complexity to the relationship between environment and growth.

In fact, the relationship between growth and environment need not take any particular shape. Several functional forms are possible. For some pollutants (in the relevant income range), the curve appears to be a constant increase (e.g., carbon dioxide emissions) or a constant decrease (e.g., bacteria in drinking water). In principle, this difference might result from variations in the turning points of the curves. Thus, it is possible that there is an inverted “U” curve for carbon dioxide emissions, but a constant increase is observed because no countries have yet reached the peak. In such cases, for all practical purposes, EKCs are irrelevant.

EKCs are not inevitable or fixed development paths. The effect of policies may often dominate over the variations described by EKCs, which is encouraging because a strict belief in EKCs would lead policymakers to accept as true that pollution increases were inevitable in the short run. In fact, fairly inexpensive measures often can limit pollution considerably, even in the short

run. It would be dangerous if policymakers were to believe that pollution and destruction of natural resources did not matter because the damage would automatically be reversed later, farther along the curve. Experience shows that “repairing” ecosystems and “replacing” natural resources is much more expensive than prevention, and in some cases, the damage is irreversible.

Recent evidence suggests that the EKC has a very flimsy statistical foundation (Stern 2004). For instance, developing countries are addressing environmental issues with a short time lag and sometimes performing better than some wealthy countries. This is mainly explained by technological change. In slower growing economies, abatement technologies can overcome the scale effect of rising income per capita; instead, in faster growing middle economies, the effect of rising income overwhelms the contribution of technological change.

A new generation of improved econometric techniques has been used to disentangle the relations between development and the environment (Deacon and Norman 2006, Madisson 2006, and Galeotti et al. 2009). The results suggest that the existence of a simple and predictable EKC is unlikely, and they point out the challenges of incorporating new global policies (for example, carbon trading) on the debate on globalization and the environment.

A related controversial hypothesis is the so-called “resource curse.” This hypothesis refers to the apparent paradox suggesting that abundance of resources is associated with (a) slow growth, (b) an enhanced risk of civil war, and (c) autocratic political regimes (these dimensions being independent) (Wick and Bulte 2009).

The paradox of the resource curse is supported by several econometrical results showing a negative association between resources and growth. The studies also show that the paradox is more likely to emerge in cases where natural resources are clustered in space and can be controlled or defended by one party (which promotes inequality due to highly skewed access to enduring resource rents) (see Sachs and Warner 2001, Neumayer 2004, Bulte et al. 2005).

Although the literature on resource curse initially focused on the impacts of natural resources on economic performance, thereafter developments have analyzed civil war likelihood and emergence of political regimes characterized by authoritarianism. For instance, Ross (2001) and Jensen and Wantchekon (2004) provide empirical evidence that links oil to autocratic regimes. In addition, in the context of resource-rich societies, resource rents provide a source of funding for the start-up costs related to initiating a civil war or rebel organization (Ross 2008).

The paradox of the resource curse has been explained by both exogenous and internal factors. Exogenous explanations include the Dutch disease, in which the significant appreciation of the real exchange rate leads to declines in productivity and competitiveness of the secondary sector (Sachs and Warner 2001); revenue volatility and a long-term declining trend of trade can also be held accountable for the poor economic performance, along with long-lived unsustainable industrial policies of protectionist barriers for import

substitution. Internal factors include corruption and rent seeking (Torvik 2002), poorly developed institutions, a crowding out of human capital accumulation, and entrepreneurship (Gylfason 2001, Mehlum et al. 2006).

With regard to internal factors, an increasing body of literature has lately drawn attention to the causality relationship between resources and growth. For instance, Brunnschweiler and Bulte (2008) question the existence of the resource curse, arguing that the causality between abundant resources, weakly developed institutions, and slow development may run the opposite direction: countries with poor institutions are unlikely to develop non-primary production sectors to reduce their dependence on resource exports. As such, the proxies employed for natural resources abundance in econometrical studies are actually endogenous to structural factors. Brunnschweiler and Bulte (2008) correct for the endogeneity and find that resource dependence does not affect economic growth, whereas resource abundance positively affects growth and institutional quality. Van der Ploeg and Poelhekke (2010) and Boyce and Emery (2011) also find that when correcting for endogeneity, there is no evidence for resource curse. Thus, these studies suggest that even if the correlation between growth and natural resource abundance is negative, this is not sufficient to conclude that there is a resource curse, since there are several institutional, political, and social variables that mediate the relationship between natural resources endowments and development paths. These variables have been influenced by historical settings, and not by the resource base itself (Mehlum et al. 2006).

Central recommendations for successful and sustainable development (World Bank 1991, Acemoglu et al. 2001, and Adhikari 2005) include formulating the correct macroeconomic policies, creating a market-friendly orientation, being open to trade, investing in people through health and education, and strengthening institutions. During the 1990s, many countries were quick to implement a market-friendly orientation and macroeconomic policies (López et al. 1999). Barriers to trade and finance were broken down; price controls and deficits were reduced. Some countries significantly increased education and health expenditures, and many countries experienced economic growth as well as declining poverty. The 1990s also showed how easily advances in some areas could evaporate into economic crisis, with an enormous price (in the form of environmental damage) of economic progress in some countries. It also became strikingly clear that corruption not only is an issue of morals but also entails enormous economic costs (López 2000).

Economic growth and environmental sustainability are complex aggregates, determined by the interplay of numerous factors; it would be foolhardy to believe that there is a deterministic relationship between the two. Technology and output composition are important, but these parameters cannot be determined directly²; they are determined endogenously in the economy. The composition of output tends to develop in certain ways that reflect factor endowment, tastes, and comparative advantages during certain

periods. However, this development results from not an iron law of physics but social behavior, which can be considerably influenced by suitable policies. Similarly, technology choices are made by economic agents and can be highly influenced by suitable policies.

The importance of a good environment for business may be greater than the availability of finance, because the latter will simply come if the conditions are appropriate. Of foremost importance for a good business environment appears to be a transparent, predictable, and reasonable legal and political structure. It must be free from corruption and exaggerated bureaucracy but also structured enough to avoid the costly uncertainty of contract enforcement. A good business environment also requires a reasonable natural environment; employees can hardly thrive or be healthy in a deteriorated environment. The distribution of environmental quality is crucial. If the living environment of the poor is so degraded (e.g., through disease or malnutrition) as to inhibit their productive development, then the economy experiences not only a direct decrease in human welfare but also a loss of productive potential.

To reconcile economic freedom, growth, and ecological constraints may require a careful blend of policy instruments to influence the composition and the technology of consumption and production (e.g., Carlsson and Lundström 2000). Research and policy experience show that certain policy instruments work better under some circumstances than in others. The choice and design of policy instruments is an important and promising area for future work. In order to deal with transboundary pollution problems such as climate change, the integration and coordination of local, national, and international initiatives is also important and timely.

Welfare and Policy Reform

Economists often assume that the well-being of an individual (i) can be expressed as a utility (U) function that depends on income, consumption, leisure, working conditions, environment, and other factors. Analogously, it is also often assumed that social welfare (W) depends on all these individual utilities, as in the social welfare function $W(U_1, \dots, U_p, \dots, U_n)$. Economists typically do not know much about the shape of these functions, but for simplicity, they are sometimes assumed to be linear in the sense that net monetary income would reflect utilities and welfare. Another common assumption is convexity of the functions, which implies that an increase in income for the poor is more important than an increase in income for the rich. Even with general welfare functions, the assumption that economists often make (and that we basically agree with) of the desirability of maximizing welfare still depends on one of several possible value judgments. Other people might prioritize an egalitarian society or a society with some other goals. For most purposes, welfare maximization is still a general goal. The welfare of future

generations may be included as well as distributional concerns and concerns about long-run sustainability.

One of the main lessons of economics is that the market mechanism is efficient at allocating resources. Economics attempts to formally illustrate this efficiency by building mathematical models of the economy. Models show that under “perfect” conditions (i.e., a market with free competition and without noncompetitive markets, public goods, or external effects), a market will automatically achieve a (Pareto) optimal outcome. This hypothesis is often referred to as the First Theorem of Welfare Economics³. Pareto optimality is an efficiency concept that implies that the economic situation of one individual can be improved only if the economic situation of another individual is worsened. Intuitively, this concept can be understood by considering the opposite: that one individual’s economic situation could be improved at no expense to anyone else. Most people would agree that this is an unnecessary deprivation. However, even this seemingly technical and neutral efficiency criterion hinges on value judgments. Making one rich person richer (with constant incomes for everyone else) may not be considered desirable. It is thus conceivable that a welfare function would decrease in some individuals’ incomes for certain values.

In general, a given economy has many possible optimal outcomes, and different starting conditions (notably, of income distribution) will give different Pareto optima. Choosing between them necessarily requires some value judgment. Criteria are needed to judge the desirability of different states of the economy that imply gains for some groups but losses for others (e.g., taxing the wealthy to help the poor). Such criteria are an expression of the social welfare function.

The Second Theorem of Welfare Economics states (under fairly restrictive conditions) that any desirable and feasible outcome of the economy that one chooses with the help of a social welfare function can be achieved as the result of a competitive economy. It implies that any outcome can be “decentralized,” that is, achieved by the market agents themselves, if the state arranges appropriate conditions (e.g., by a lump-sum redistribution of the initial endowment). It means reallocating money—taking money from some individuals and giving it to others—but otherwise leaving the economy and its mechanisms intact. Such redistributions do not always work in practice (partly because taxes and subsidies influence people’s behavior), but in some cases, policy instruments can decentralize the outcome⁴.

A real economy could not be a “pure market” in the absolute sense of having no state interference. The mechanisms that make the market work (e.g., the definition and enforcement of property rights and civil laws that govern contract enforcement) are public goods that have to be provided for by a public body. These and other necessary mechanisms, such as the maintenance of social order and defense, are also costly activities that make at least some level of taxation an inevitable feature of the economy. Taxation requires resources, and its implementation tends to distort the price signals

of the market, which modifies the optimal properties that can be derived in a simple and abstract market with no outside interference. As soon as one aspect or area (such as taxation) deviates from the simple textbook model of the “perfect market economy,” the conclusions and recommendations from that model may no longer apply. Policies or outcomes that would be best, given that some imperfections already exist, are called “second- best” by economists. Economists’ ability to analyze the optimality of second-best situations is limited; however, it is commonplace (and perhaps, on the whole, reasonable) to believe that at least some of the “first-best” efficiency properties of market solutions remain valid; an instrument is “first-best” when it would be optimal under some set of ideal (often unrealistic) market conditions, which often implies that the instrument is not optimal in the real (imperfect) world.

The virtual collapse of many formerly planned economies that attempted to rush the transition to the free market without due attention to building the necessary institutions is a good, although unfortunate, illustration of two important facts:

- Economies with an excessive degree of state intervention fail severely in attaining efficiency.
- Economies with an excessively free and unregulated market may fail abysmally on both efficiency and social issues.

Ironically, some economies (e.g., the Russian Federation and other countries of the former Soviet Union) rapidly moved from what could be characterized as “excessive state” to insufficient or perhaps inadequate state institutions. They learned the hard way that the market is a social institution and one that requires considerable enforcement from a state strong enough to defend property rights and uphold a necessary degree of trust and impartiality in civil law if entrepreneurs are to feel comfortable investing in the economy.

In the economic area, a general absence of rules would lead to stagnation; in the ecological area, it could lead to expensive abuse of originally productive resources. However, the relative importance of institutions may vary depending on such factors as cultural norms and the characteristics of technology. Less political enforcement may be needed in extremely structured cultures with a tradition of a high work ethic and an emphasis on honesty; feedback from economic outcomes and political institutions gradually changes social norms. The institutions needed in an economy where technological progress is slow and labor-intensive will be distinct from those needed in an economy where technological progress is rapid and capital-intensive. Other important factors are the market structure, the size and openness of the economy, and issues related to risk and information asymmetries.

Over the past few decades, the market mechanism has shown its many strengths in real-world economies. The exceptions are nevertheless important, especially as applied to the management of natural resources and ecosystems.

This area is characterized by externalities, public goods, common pool resources, imperfect foresight, and other types of market failure. The concern for social welfare implies a special focus on the poor, which in turn tends to imply that risk and uncertainty are given greater weight than maximizing expected return. Variation in income is not an acceptable risk for people who live in danger of starvation; it is far from the “first-best” world in which all marginal costs and utilities are equalized.

One common problem for the analysis of the cost-effectiveness of environmental policies is that the process of policy reform (e.g., redesigning taxes) changes the relative prices and incomes in a way that makes comparisons with the original state difficult. A successful process of policy reform may well entail taking temporary steps that are not efficient (Guesnerie 1977). These issues are particularly well illustrated in developing countries, which have large disparities in income (and thus in the marginal utility of money). One empirical study that attempted to design optimal tax reforms for India used a technique referred to as the inverse optimum, which entails determining which set of welfare weights would have made the observed state of the economy an optimum (Ahmad and Stern 1984). If the calculated weights do not match the welfare weights that can be assumed for the decisionmaker (especially if they contain weights that are blatantly impossible—such as negative weights for some groups), then tax reform that increases welfare must be possible.

Market Failure

The conditions under which the welfare theorems hold (i.e., a “perfect market”) are convenient analytical abstractions that provide a starting point for economic analysis. A situation in which those theorems fail to hold is called a *market failure* and is very common. In this section, we focus on four kinds of failure: *noncompetitive markets*, *external effects*, *public goods* and *common pool resources*.

The idea of a market is that people engage in mutually beneficial trade, and to do so, they must have clear ownership rights and information. In a perfect market, every good and resource has an owner and a price, and the agents have full information of the options available to them. Production and consumption technologies are characterized by the absence of indivisibilities and increasing returns to scale (i.e., the rate at which output changes as the quantities of all inputs are varied)—or, more formally, by the absence of nonconvexities in production and consumption sets. This seemingly technical explanation is illustrated in Figure 2-2.

The intuitive meaning of convexity is that you can combine goods (or production inputs) at will within the set of possibilities. This assumption is often taken for granted, and understanding how its absence can render optimality unachievable is simple: suppose that you want to buy bread and cheese, but the smallest package of each is so large that you would have to spend your

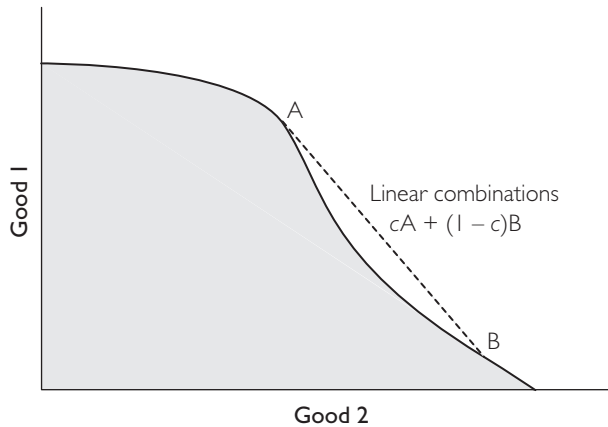


FIGURE 2-2 Convex Combinations in Consumption Space

Note: Input or consumption bundles *A* and *B* are both feasible, but combinations such as one-half of *A* plus one-half of *B* (or more general linear combinations) are not attainable because of the nonconvexity shown.

entire budget on only one item. Although this example sounds trivial, indivisibilities are sufficient to prevent the attainment of optimality. Similarly, the high minimum-efficient scales of refineries or steel mills cause problems in small countries; small plants are not cost-efficient, big plants may have insufficient demand, and no plant means that the country will be 100% dependent on imports. The same applies to infrastructure investments such as bridges, health programs, and railways. A country may desperately need infrastructure but find the available systems excessively large and prohibitively expensive. The absence of intermediate-scale solutions implies constrained choice, and a decentralized market will not lead to an optimum. Such a nonconvexity may be created by increasing returns to scale.

Noncompetitive markets such as monopolies and oligopolies are a kind of market failure that usually leads to excessively low production volume being sold at too high a price. This situation diverges from the optimum because an increase in production would be possible and would be valued more highly than the additional cost. The existence of monopolies in the economy is partly related to underlying cost structures, such as increasing returns to scale. They are generally accepted as something that policymakers should regulate.

Typically, external effects also create this kind of nonconvexity. Consider two goods, *A* and *B*, that have strong negative externalities between them. You can have *A* or *B*, but obtaining both is difficult. Similarly, by definition, public goods imply that everyone has to consume the same quantity: there can be one state of the economy in which everyone gets none of the public good and there can be a state in which everyone gets a quantity *z* of the same good, but there cannot be a state in which some people get none and some get *z*.

This restriction, too, means that the decentralized market will not lead to an optimum. Other causes of nonconvexities include common pool resources, congestion, and joint production.

The simplest economic models are deterministic and atemporal, but the real-world economy takes place in real time, and its outcomes are stochastic; the best one can hope for is to know their probability distribution. In general models of the economy, a “space” of goods—each of which is labeled by a probability and a date—can be defined. In such a model, there can be many more sources of nonconvexity, including varying degrees of myopia (near-sightedness), uncertainty, risk aversion among the agents of the economy, and transaction costs.

For a market to be perfect, all property rights must be fully allocated. No externalities, public goods, or other nonconvexities should exist. There also must be markets for all goods and resources, including, most importantly, future goods (i.e., future markets) and full information about all these markets. The mere existence of a market failure does not automatically warrant the implementation of a given policy because the costs of market failures must be weighed against the potential for “policy failures.” This comparison must be carried out within the specific context of general policymaking in the economy to be studied (see Chapter 3).

Externalities

An externality can be defined in different ways, but it typically is an unintended and uncompensated side effect of one person’s or firm’s activities on another. Good examples are the health effects of smoke emissions from vehicles, factories, and cigarettes. These side effects occur because of a technical interdependence in consumption or production (see Box 2-1, Definition of Externality). They are not intentional damage per se, and they typically are difficult to avoid. Note that this interdependence must also be a nonmarket dependence to qualify as an externality. If many people are lined up to buy a good (e.g., medicine or water), the price of which consequently increases, then the effect (commonly referred to as a pecuniary externality) is not an external effect, because it is perpetrated through the market mechanism. If most people in an area take antimalaria drugs and this action ultimately decreases the number of malaria mosquitoes (and thus the malaria risk to individuals who do not take tablets), then people who do not take tablets are the unintended beneficiaries of an external effect.

Another approach to defining externality is to suppose that ownership rights or markets are missing for the particular resource in question. For example, if there were private ownership of the air, then people would have to buy the right to pollute it with smoke, and passive smoking would be internalized through the market. Practical barriers to the establishment of such rights and markets, however, are likely.

BOX 2-1 Definition of Externality

A general definition of an externality is the existence of some variable that enters into the utility or production function of an agent (an individual or a firm i) in the economy, although it is controlled by another agent (j) who does not take effects on i into account and does not pay compensation. For the case of utilities, the utility (U) of individual i depends not only on his own consumption but also on the consumption (or some other variable) of another individual j :

$$U_i = U_i(x_i, x_j) \quad (2-1)$$

One example of this function is second-hand cigarette smoke.

When many agents affect each other, the function can become quite complicated. However, any information about the nature of these functions can be included. For example, if a number of agents ($j = 1, \dots, m$) emit smoke (s_j) and this emitted smoke is perfectly mixed in the atmosphere, then it is the sum of all the smoke ($S = \sum s_j$) that affects utility, allowing a simplification of the equation:

$$U_i = U_i(x_i, s_1, s_2, \dots, s_m) = U_i(x_i, S) \quad (2-2)$$

Externality is perhaps the most basic concept in environmental economics. Long recognized as a problem, it originally was seen as a minor one. The classical economists wrote of the soot from factories in Manchester and Liverpool, England, that dirtied the laundry hanging on the line and of the bees who pollinated neighboring farms' orchards. Sometimes the beauty of a rose garden, enjoyed not only by the owner but also passers-by, was used as an example. These examples clarify the issue but also border on triviality. However, the environmental issues that confront us today—contaminated drinking water, smog in developing-world cities, destruction of the ozone layer, acid rain, and global warming—are far from trivial. Unfortunately, the dependencies might be long-ranging, which makes defining property rights or negotiating difficult. Examples include cases in which the polluter and victim are separated by long distances (e.g., the effects of soil erosion on coral reefs) or in time (e.g., the risks posed to future generations by nuclear waste).

The environment and the various services provided by ecosystems enter into ordinary production and consumption in ways similar to those of other inputs. Thus, they should be included in economic accounts at corporate and national levels. They often are not, because there is no "owner" or because the environment has the characteristics of common property or a public good. The absence of property rights is related to scarcity. Classical writers such as Marx noted that water might have great "use value" but little "exchange value" when plentiful. Without water, there would be no production and no life, and thus, in a sense, water is "infinitely valuable." In countries where it is abundant, it is practically free. Similar arguments apply to many natural, environmental,

and ecosystem resources. Oxygen, phosphorus, DNA, chlorophyll, iron, and biodiversity—to name only a few—are individually infinitely valuable.

Trying to estimate the total value of global ecosystems is pointless because the value is infinite. Only marginal changes can be studied. History shows that property rights and market values appear only when use value is coupled with scarcity. Thus, the existence of external effects is intimately tied to the absence of markets, and this absence, in turn, is the result of a certain social and historic condition. In fact, the absence of property rights or of markets is an alternative way of defining externalities. At one time, there were no rights to land anywhere; today, most land is claimed, and agents are staking out rights to radio waves, geostationary parking slots, genetic codes, and even property lots on the moon. (Property rights are discussed further in Chapter 3.)

Externalities are commonly distinguished as depletable or nondepletable. The manure from horses is a depletable externality because if one person takes it, another cannot. However, the odor of horse manure is a nondepletable externality because one person's exposure does not reduce the exposure to others. (This concept also applies to congestion and is essentially the same as the nonrivalry of public goods.)

The effect of externalities on resource allocation is illustrated in Figure 2-3 for production externalities. It shows the usual market analysis for a certain product, with supply determined by ordinary private production costs (MC_p). Market equilibrium is determined by the intersection of demand and supply curves (Q_m, P_m). If each unit of production gives rise to a certain (for the sake of simplicity, constant) external effect, then there is an extra cost to society (MC_e) that is not borne by the producer. This damage would be measured as the sum of the decreases in utility due to the external effect for all individuals or firms affected. For individuals, it would be $MC_e = e \sum_j U_{ji}'$ (i.e., the sum of marginal disutilities for all j of acts carried out by i). If internalized, it would be a social marginal cost of production (MC_s), $MC_p + MC_e$. The intersection of this curve with the demand curve gives the social optimum (Q^*, P^*). The analysis is often more complicated, because several production methods give rise to different quantities of externalities. Also, the same physical emissions might cause different amounts of damage, for instance, depending on the location of the pollution source.

Public Goods and Club Goods

The most classical of market failures is the failure to provide public goods that are not consumed by individuals but enjoyed by all or most citizens as a whole. Provision of public goods is one of the fundamental reasons for government. Many natural resources or ecosystem services are at least to some extent “public” in this sense. In order to discuss the implications for policy design, several related kinds of public good must be distinguished. Pure public goods are characterized

by nonexcludability (if a public good is provided for some individuals, others cannot be excluded; e.g., national defense) and nonrivalry (the enjoyment of a public good by one individual in no way reduces its availability to others; e.g., television broadcasts). According to public economics, the market alone cannot allocate resources optimally between public goods and private goods; nonexcludability directly invalidates the use of the price mechanism for resource allocation. Affected public goods include defense, law and order, education, and health—even “a clean environment,” which can be seen as a kind of public good (and pollution, which might be considered a “public bad”).

Because the market does not provide public goods (except in small quantities or special cases, such as when public goods are provided through charity or some form of sponsoring), the state or some other political body is the origin of collective action that produces them. The most common starting point for a discussion of the optimal provision of public goods is the Samuelson rule, which declares that the social value of a good is equivalent to the combined willingness to pay of (or utility to) all the consumers of that good (Samuelson 1954, 1955) (see Box 2-2).

This rule is similar to the one discussed for externalities. If agent i 's consumption somehow leads to a benefit for j , then this extra utility will not automatically be considered by i —but it should be if social welfare is to be maximized. If everyone enjoys a given public good equally, then the benefit for society is the sum of all the individual utilities. This is an abstract, “first-best”

BOX 2-2 Derivation of the Samuelson Rule for the Provision of Public Goods

To determine the optimal provision of a pure public good, the utility for household 1 (U_1) is maximized as $U_1(x_1, G)$, where x_1 is household 1's consumption of a private good and G its consumption of the public good, subject to a minimum level of utility for other households (j) and a production function (F). It can be achieved by maximizing the Lagrangean function L :

$$\max L = U_1(x_1, G) + \sum_{j=2, \dots, n} \mu_j [U_j(x_j, G) - U_{j0}] - \lambda F(x, G) \quad (2-3)$$

Taking partial derivatives and solving gives the Samuelson rule:

$$\sum_j U'_{jG} / U'_{jx} = F'_G / F'_x \quad (2-4)$$

where U'_{jx} is the marginal utility for household j of private good x ; the other derivatives are defined analogously. The provision of public goods is optimal when the sum (over all households) of the marginal rates of substitution between the public good and any private good is equal to the marginal rate of transformation (in production) between them. If x is the numeraire, then the expression in Equation 2-4 is simply the price of the public good. If lump-sum taxation is not available, then more restrictions are taken into account, and typically, additional terms (such as welfare weights for individuals with different incomes) would be added to the left side of Equation 2-4.

rule for the provision of public goods in a world where, among other conditions, individually differentiated lump-sum taxes are possible, so income distribution is not a variable that must be considered⁵.

In more realistic models, the political and tax systems present various difficulties. For instance, the optimal “second-best” provision of public goods is lower than the first-best because of the cost of raising the tax needed to finance the public goods or the effect of the tax on labor supply. The public sector may have several goals, including an even distribution of income. Such a goal typically complicates models if the distribution of consumption patterns is different for the rich and the poor. When income distribution goals cannot be met by using taxes and subsidies, and if the public good in question is particularly attractive to poor people, then it may be optimal to increase its provision. Similarly, the optimal tax structure for different goods may reflect a mixture of goals: A negative externality related to a particular good is a factor that leads to a higher tax, whereas consumption of that particular good by the poor is usually an argument for a lower tax. The construction of optimal taxes is complex, and the results depend on the model. With some forms of income tax, this contradiction need not arise.

Two other categories of goods are closely related to public goods but often have some degree of congestion (i.e., “costs” are related to the use of a good by many people) and rivalry in use. *Impure public goods* include such seemingly classical public goods as parks and roads, but the utility of one user typically is reduced by an increase in the number of other users. *Club goods* (sometimes called mixed goods), categorized between private goods and public goods, can be consumed by many individuals without diminishing the consumption of others (e.g., a movie). However, exclusion (of nonmembers) is possible.

For both of these categories, the first-best Samuelson rule still holds in the sense of Equation 2-4: The club good should be supplied in such a quantity that its price corresponds to the sum of the n club members’ marginal willingness to pay. Several interesting issues remain to be solved, such as the number and size of the clubs (n). As long as exclusion (of those unwilling to pay the price of the club good) is possible, the decentralized market economy can solve this problem and provide club goods in an efficient manner (Buchanan 1965). Similarly, if people are willing to make trade-offs between the quality of local public goods (i.e., goods that are public to a community or municipality) and taxes, then different local communities could offer mixed public goods of different quality and citizens will base their location decisions on the choice between paying higher taxes with more “free” services or lower taxes with fewer such services (Tiebout 1956).

Common Pool Resources

The ecosystem functions of the natural environment and various natural resources not only enter into the utility functions but also commonly are

inputs into production. When these resources have some difficulty with the definition of private property rights—they may be referred to as *common pool resources* (or “commons”). Common pool resources are usually characterized by costly exclusion, and there is typically rivalry in use: The resource is common, but the goods produced are private (Ostrom et al. 1999). If it is technically impossible or too costly to deny open access to an environmental resource, market allocation is likely to be inefficient (see Box 2-3). In such cases, individuals over-exploit the resource relative to what is best. Fishing grounds on the oceans provide the best-known examples of open-access commons. More fish caught by one party implies less fish for others. Each fisher therefore has incentive to increase the fishing effort to capture the rents. If each fisher ignores how his actions affect others, total effort expended exceeds the socially optimal. Since the scarcity value is ignored, overfishing and depletion of the fishing stock can be the results.

Some common pool resources have no formal owner, but typically some form of ownership control is exercised, collectively or by private individuals. In several respects, these resources act like local public goods or club goods except that they are generally more a source of production inputs than goods for direct consumption.

In the past, the concept of common pool resources has been confused with that of “free access” resources; the difference is similar to that between free fishing in the ocean and participation in a local river fishery strictly

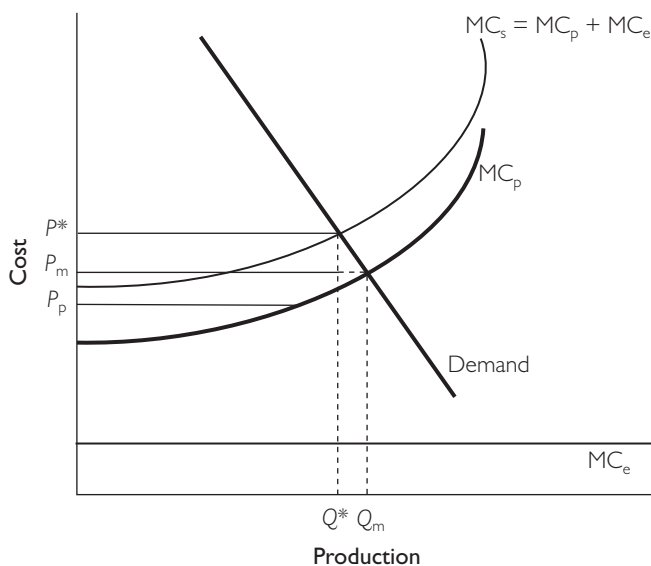


FIGURE 2-3 Externalities and Their Effect on Markets

Note: P = price; Q = quantity; MC = marginal cost. Asterisk indicates optimum value. Subscripts m, p, s, and e represent market, private, social, and emissions, respectively.

BOX 2-3 Non-Exclusion and the Commons

To illustrate the inefficiencies that arise when it is technically impossible or too costly to deny open access to an environmental resource let us consider the problem by n producers that have access to a common resource. Total harvest (for instance, fish or timber) from the commons is determined by total effort expended $\hat{x}$ by all the users. Let the productive function be $y = F(\hat{x})$, where $F'(\hat{x}) > 0$ and $F''(\hat{x}) < 0$. We can assume that harvest is sold on a competitive world market at a price p per unit. Each producer's share on the total harvest is determined by his effort x relative to total effort: $(x/\hat{x}) * F(\hat{x})$, where $\bar{x}$ is the total effort of the remaining producers and w is the opportunity cost of harvest effort.

The social optimum level of effort is determined by maximizing the profits of all the users:

$$\Pi(\hat{x}) = pF(\hat{x}) - w\hat{x} \quad (2-5)$$

The socially optimal level of harvest is determined by the FOC which equates the value of the marginal product to the opportunity cost of effort:

$$pF'(\hat{x}) = w \quad (2-6)$$

In contrast, the market outcome is set by the profit maximizing behavior of each user:

$$\pi(x) = p \frac{x}{x + \bar{x}} F(\hat{x}) - wx \quad (2-7)$$

The FOC for this problem is given by:

$$p \left[\frac{x}{\hat{x}} F'(\hat{x}) + \frac{\bar{x}}{\hat{x}} \left(\frac{F(\hat{x})}{\hat{x}} \right) \right] = w \quad (2-8)$$

The marginal benefits are a weighted average of the marginal product $F'(\hat{x})$ and the average product $F(\hat{x})/\hat{x}$. The weighting scheme depends on the number of users. If we assume that all producers are identical (2-8) becomes:

$$p \left[\frac{1}{n} F'(\hat{x}) + \frac{(n-1)}{n} \left(\frac{F(\hat{x})}{\hat{x}} \right) \right] = w \quad (2-9)$$

If we consider the case of a unique producer, the market optimum corresponds with the social optimum. The monopolist producer selects the social optimum level of harvest because he captures the scarcity rents associated with the commons. But if the number of producers is very large, each producer equates the value of the average product with the marginal cost and the economic profits disappear toward zero. This implies that producers using the commons expend too much effort; they all ignore the scarcity value of the resource. This is the so-called "tragedy of the commons." While the arithmetic is impeccable, some additional thought clearly is needed when it comes to the realism of the behavior and institutions involved. It is part of Elinor Ostrom's greatness that she has so consistently transcended political science, economics, and anthropology to explore this, using everything from field studies to game theory and behavioral theory.

regulated by village elders. However, the exclusion of outsiders is an important feature of successful common pool resources. Excluding members (or temporarily limiting harvests) is also practiced. The main difference between common pool resources and local impure public goods is that the size of the resource is determined by nature: Overharvesting can reduce the productivity of the resource, and the resource is depletable. Good management may enhance the productivity of the resource, but its size cannot be decided arbitrarily. Resources that have certain features (such as fluidity, mobility, and extensiveness) are often used by many people. These resources are referred to as common pool resources. If these resources are managed jointly, this is common property resource (CPR) management, an important policy instrument (see Chapter 8).

Congestion

Public goods are rarely “pure”; usually, some rivalry or congestion exists. For example, at low levels of use, a highway is a pure public good that gives utility to any user, irrespective of others’ use. With increasing levels of use, congestion increases, and the utility derived by each motorist is a decreasing function of the number of other motorists; each additional car implies a negative externality on all other motorists. This effect is illustrated for traffic congestion in Figure 2-4, in which the social marginal cost (MC_s), which includes the sum of the time costs caused by the delays that this motorist imposes on all other motorists as a result of increased congestion⁶, diverges more and more from the private marginal cost (MC_p), which consists only of fuel and vehicle-related costs plus the motorist’s own time. The first-best optimum use of a given highway would be at an optimal traffic volume (Q^*), which could be obtained by charging an extra road charge or tax⁷ (P^*) (Figure 2-4).

Moving from congested equilibrium (Q_c) to Q^* implies a collective gain in welfare (W) by the motorists. The trouble with this solution is that the motorists have to pay the entire rent⁸ ($R = P^*Q^*$) to get W . Therefore, they may be collectively better off in the congested equilibrium Q_c . This is a dilemma for policymakers, who need to worry about losing votes from particular groups and might therefore be tempted to spend revenue on policies that benefit important groups of voters. Figure 2-4 can be used to identify not only the total costs to motorists—it can be broken down to show their tax payments as well as the time gain for those motorists who continue to drive after congestion is reduced and the loss in consumer surplus for those who stop driving. If all the external costs were congestion costs and if only motorists were affected, then perhaps a case could be made that the state should refund at least such a share of the taxes as to leave the motorists no worse off in Q^* than in Q_c . However, in the real world, where congestion-related externalities (e.g., exhaust) also affect nonmotorists and residents in different parts of a

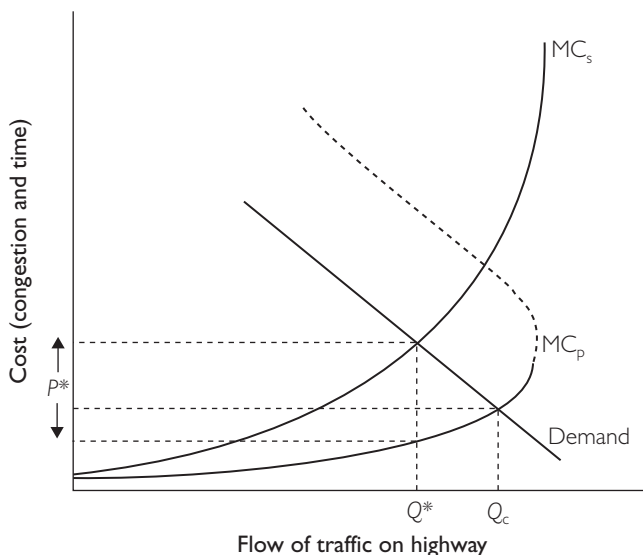


FIGURE 2-4 Congestion Pricing for Highway Traffic

Notes: MC_s = social marginal cost; P^* = extra road charge or tax; MC_p = private marginal cost (fuel and vehicle-related costs, plus the motorist's own time); Q^* = optimal traffic volume; Q_c = congested equilibrium.

town differentially, the issue becomes more complicated (for alternative views on earmarking road funds, see Chapter 17). Some participants in the debate consider that all road charges “should” be returned in the form of road investments. In several countries, including the United States, it is mandatory to use road funds in this way. This approach is not at all in accordance with a welfare theoretic analysis, but it still may carry political weight because motorists tend to be a fairly vociferous lobby.

Congestion depends on the relationship between flow and density. If congestion is severe—called hypercongestion—then the marginal damage curve “bends backward,” and congestion costs increase. In some cases, motorists gain from a charge that reduces congestion and increases traffic flow. When congestion grinds traffic to a halt, almost any measure that increases traffic flow is better than nothing (see Part IV).

Many other resources with free access—such as fishing—could be characterized by and analyzed in terms of congestion. The analogy is not only that each fisher increases the costs of others but also in the fact that policies to deal with congestion may be beneficial at the aggregate level but still be difficult to implement because they do actually risk hurting at least some influential groups.

Asymmetric Information and Uncertainty

One of the most important and often neglected “goods” in the economy is information. Information is vital for economics transactions and for market function. Simple economic models assume full and free information, but technical information is produced by hard work and usually is neither free nor symmetrically distributed. Economists have come to realize that lack of this vital tool—particularly in conjunction with risk and uncertainty—has real effects on resource allocation and on welfare.

Information is difficult to protect with property rights, although patents partially fulfill this function⁹. It is difficult to exclude noncustomers from benefiting from information; therefore, information tends to be undersupplied in a pure market economy, a well-established motivation for public funding of research, dissemination, and teaching.

Information asymmetry (or “agency problems”) is a situation in which some agents (e.g., employees, firms, or regulators) have different information. With risk, information asymmetry is an important underlying cause of different kinds of nonoptimality in the economy. It may be directly relevant to environmental issues (e.g., the risk of chemical accidents or oil spills) or indirectly relevant, because it causes distortions (e.g., in the factor or insurance market) that may interact with environmental externalities. Information asymmetry also is likely to turn up in the design and implementation of policy instruments themselves, because firms typically do not automatically disclose the whole truth to their regulators.

Most productive activities require collaboration between agents. Advantages to scale, specialization, and other factors such as level of risk aversion result in many people contributing to one final outcome. This introduces the problem of writing contracts to define the relative contributions of the agents and their remuneration. For example, consider poor farm laborers who work for a landlord. If the harvest (outcome) depends on weather (a random variable), then the remuneration of at least one party to the contract must be stochastic. If the laborers cannot tolerate large uncertainty because they live close to the poverty line, then they may be better off when the landlord (capital owner) bears the risk and pays the laborer (agent) a fixed but low wage. This potentially good arrangement gives security to the laborer and a higher long-run (expected) payoff to the more risk-neutral landlord. The main trouble is that the incentives for the laborer to maximize harvests are gone, and the landowner may have to spend large amounts of resources in monitoring (Cheung 1969; Stiglitz 1974). This situation is characteristic of many rural settings and is necessary to incorporate in analyses of pollution and natural resource degradation in poor countries (e.g., Binswanger and Rosenzweig 1986).

In the typical principal-agent problem, in which the interests of the principal (shareholder) and agent (manager) differ, the outcome for the principal depends on the effort of the agent as well as stochastic factors (i.e., random

variables such as weather), so the principal cannot in any simple way isolate the effect of the largely unobservable effort by the agent. This information asymmetry after the contract is signed is called a moral hazard. (It also occurs when an agent who is insured against risks takes less care in protecting himself or herself because he or she does not bear the full costs of a mishap. In such a case, the insurance company must determine whether damage was incurred as a result of carelessness or random events of the kind that insurance is intended to cover.)

A related category of problems is referred to as adverse selection, a situation in which the information asymmetry concerns not the agent's behavior but the agent's characteristics when the contract is signed (see Chapter 10). If the agent has private information about his or her productivity, health, or other characteristics that are relevant to the outcome and that the principal cannot observe, then first-best contracts are unobtainable. For example, healthy individuals are least likely to purchase insurance policies. Therefore, policies priced based on average health statistics will be unprofitable for the insurance company if their customers are adversely selected (i.e., their propensity to illness is higher than the average), so the insurer charges a higher premium, which reinforces the self-selection of policy buyers. It tends to indicate that a healthy person cannot purchase reasonably priced insurance, so at least some healthy people will not insure themselves at all—an undesirable state and one that may affect their behavior in other areas of relevance to the environment. Still, a healthy person who is extremely risk-averse might purchase insurance anyway.

Adverse selection also can apply to the quality of labor, products, and services. If a buyer or employer cannot distinguish between the characteristics of products or employees, then market price and quality supplied will tend to be lower than optimal. The potential top-quality producers will not be fully rewarded if buyers cannot distinguish between low-quality and high-quality products, and either high quality will not be produced at all or significant resources will be spent on signaling (i.e., methods that are used to indicate quality, such as advertising, warranties, and educational titles). In the labor market, some economists argue that education is mainly a way of signaling relative performance rather than of preparing for actual productive tasks. The same applies for the provision of environmental information (see Chapter 8).

Some cases may allow partial solutions—for instance, based on using information from the correlation of harvests between adjacent plots, the history of agents to be insured, or signaling for quality. Sometimes the information asymmetry (or the cost of monitoring) is so severe that the principal would like to make the agent the owner of the residual uncertainty. This arrangement implies the use of a completely different form of contract, such as rental, in which a laborer (agent) rents land (capital) from a landowner (principal). The laborer reaps the rewards for any extra effort and thus has incentive to increase harvest (outcome). The landowner receives a steady stream of profits

(although their expected value is lower than what could be obtained if the landowner were to assume the risks and could solve the problem of monitoring and could provide adequate incentives to the agents). In this situation, however, the laborer must bear the economic risks of variable factors such as adverse weather.

One solution in farming, business, and insurance is some form of risk sharing between the principal and the agent: crop sharing in agriculture, stock options for business executives, and deductibles in insurance contracts. Risk and reward can be shared in many ways, as illustrated for farming in Figure 2-5. Because of the costs of monitoring and differences in risk aversion, contracts commonly deviate from the simple neoclassical models in which wages equal marginal productivity. Similarly, rules such as equating the environmental tax to the sum of marginal damages may have to be modified (Laffont 1994b).

Although Figure 2-5 illustrates the relationships between a landowner and a farm laborer, they could equally well be between a capital owner and an entrepreneur or a regulator and a polluter. There are similar trade-offs: agents can reduce risk, but at a cost (payoff will be less than marginal productivity). Similarly, if principals assume risk, then they make more money but also must handle monitoring and incentives to agents.

In the case of oil spills from ships, full liability would be the counterpart to a pure land rental in Figure 2-5: the agent bears the full risk. However,

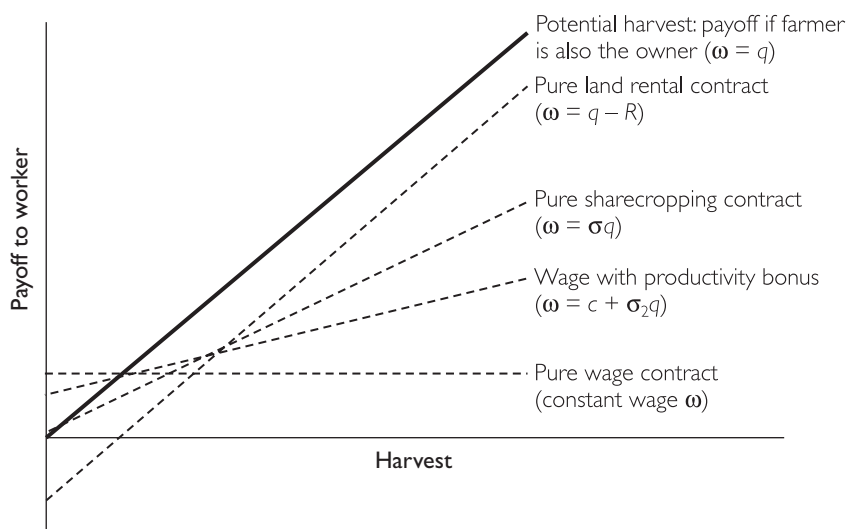


FIGURE 2-5 Sharecropping and Other Contracts for Dealing with Risk and Uncertainty

Notes: σ = share parameter (or coefficient); ω = wage level. Solid line shows potential harvest (q) if the farmer works optimally; if the farmer is also the owner, then q = payoff (ω). Dotted lines signify payoffs to workers under four contractual schemes (e.g., a worker who pays rent $[R]$ for the land receives a payoff, $\omega = q - R$).

this situation is unacceptable for two reasons. First, oil spills are emitted by multiple mobile sources, so to some extent, there is a nonpoint-source pollution problem, and it is difficult to prove that a particular ship is responsible. Second, the potential outcome (damages) is such a large economic cost that the agent may not be able to pay. Requiring full insurance would be the closest correlation to the pure wage contract in Figure 2-5, but such insurance is not available for reasons of moral hazard. If provided, it would remove all incentives for the ship to be careful. Thus, other combinations of instruments—regulation, partial insurance, and liability—are typically observed. For example, large individual ships or particularly sensitive bodies of water could be monitored, and techniques such as satellite surveillance and “fingerprinting” of oil loads with chemical or radioactive markers could be useful (e.g., Florens and Foucher 1999; Gottinger 1998). Multiple principal–agent relationships are involved: the environmental protection agency is a principal vis-à-vis its agents, the shipping companies; the companies are principals vis-à-vis their agents, the captains; and the captains are principals who must monitor their agents, the officers and sailors.

Potential problems of adverse selection and moral hazard—or inferior quality and carelessness—exist at each level. The market tends to under-supply insurance and product quality in general, producing goods of inferior quality (“lemons”). Buyers cannot separate out the lemons, and so their willingness to pay for the whole product group decreases—reinforcing the tendency for producers to supply lower quality (Akerlof 1970). Undersupply of quality is most likely to apply to environmental goods and services because of their public good characteristics. This situation in itself may be cause for policy intervention to protect safety and ensure the well-behaved function of markets (or industries, such as shipping).

Many of the problems of adverse selection and moral hazard are presumably exacerbated in developing countries, where high levels of uncertainty and greater risks of undersupplied quality and insurance exist. Low-quality product and labor markets are often the subject of casual empiricism in developing countries. This negative association can be partly remedied by increasing investment in signaling.

Adverse selection and moral hazard also have effects on insurance. The closer an individual is to the poverty line, the more detrimental variations can be, and the greater the need for insurance or other mechanisms for smoothing consumption over time (e.g., banking services). However, the administrative and informational costs of supplying these services are prohibitively high in poor countries, as are the risks associated with moral hazard and adverse selection. (Costs and risks are a factor of not only poverty but also phenomena such as mobility, morbidity, lack of public monitoring, and, in some cases, cultural heterogeneity (e.g., Narayan et al. 2000). Thus, insurance is under-supplied, which may have important environmental effects if the variation of natural systems can be controlled by technological innovations (e.g.,

irrigation, fertilizers, and pesticides) that imply health and ecological costs or risks. (Fluctuations may fill an important ecological role but still be quite detrimental to humans.) Consequently, environmentally hazardous technology can substitute for (unavailable) insurance, leading to the overuse of dangerous pesticides to reduce the uncertainty caused by pests, for instance. Similarly, large herds may be a nonmonetary “savings account” in response to the unavailability of conventional banking services. This response can be quite rational for individuals but severely detrimental for society (see Chapter 26).

Intertemporal Resource Allocation

The current population is responsible for extensive and expanding environmental damages and depletion of natural resources. Many waste streams are disposed in amounts that exceed the natural capacities of ecosystems to assimilate them. The resulting environmental damages are significant, irreversible, long-term and asymmetrically distributed over time: the net benefits accrue now and the net costs accrue in the future. On the other hand, all the economic activity ultimately depends on the environmental resource base. This resource base is finite, and its imprudent use may irreversibly reduce the capacity for generating natural assets in the future. As the extent of the damages and depletion increases, the question becomes more urgent: should this generation care about its actions that result in a degraded environment in the distant future? If we do care, what should be done?

Of particular importance is the need for reforms that improve the signals that are received by resource users, but these reforms should also ensure that the transfers of assets to future generations are adequate. Hence, improvements in the long-term allocation could be pursued by internalizing environmental externalities, but while environmental externalities are no doubt a problem, suboptimal allocations due to inappropriate distribution between generations cannot be solved solely by correcting externalities.

Intertemporal general equilibrium models incorporating overlapping generations and resource constraints demonstrate that the efficient allocation of resources is a function of intergenerational distribution and that may result in unacceptably low living standards for future generations under certain economic conditions and welfare criteria (Dasgupta and Heal 1979). Many economists, as well as environmentalists, have argued that discounting the benefits received and costs borne by future generations in project analysis is contradictory with a concern of sustainability. Lower discount rates would give greater weight to the future: Using rates different than market rates might result in inefficient use of capital, but under certain circumstances this may still be optimal (see for instance Gollier 2010).

Markets themselves do not provide for intergenerational equity any more than they provide for intragenerational equity. Capital markets might fail to

both maximize the welfare of the current generation given its consumption time preferences and meet the current generation's goal of transferring assets to future generations for at least two reasons. First, private redistribution has public good features. Second, savers are probably unable to control whether they are receiving a return from investments which will be transferred to future generations or from investments which are depleting the assets that might have been transferred to future generations. Thus, monitoring and guidance mechanisms are needed to supplement capital markets (Norgaard 1992).

Limits to Rationality and Behavioral Economics

Over the last decades the standard neoclassical model, based largely on the assumption of expected utility maximization, has come under increasing criticism from both outside and inside the economics profession due to the large number of empirical anomalies that the standard model fails to explain. In contrast to the argument that rational choice theory provides the necessary elements for guiding decisions, behavioral economists have come to introduce psychology into economics to provide more insights and guide the foundations of new theories of choice. The field of behavioral economics has developed its own taxonomy of behavioral biases (as a parallel to the idea of market failure), classifying situations where people fail to behave as predicted by rational choice theory in three categories (see Shogren and Taylor 2008): *bounded rationality*, *bounded willpower*, and *bounded self-interest*. Bounded rationality implies that people do not have unlimited abilities to process all the information needed to make rational choices. Instead, they use rules of thumb and shortcuts. Bounded willpower reflects the idea that people sometimes lack self-control: we consume too much, save too little, make rash decisions, and so on. Finally, bounded self-interest captures the fact that people can be selfless. They have social preferences for emotive ideas like reciprocity, altruism, paternalism, and aversion to inequality.

Behavioral research has not only fundamentally changed the field of economics at the conceptual level but also broadened our methodological repertoire by adding an experimental basis. A laboratory experiment in economics attempts to create and study a small-scale microeconomic environment to uncover systematic relationships between individual preferences, institutional parameters, and outcomes. The laboratory set-up is meant to capture the essence of the issue under consideration. This requires the experimenter to make some decisions about "institutional parameters," as for example, the degree of information provided in the instructions, the way the information is presented to participants, and the communication allowed among participants. Participants are provided with information about the players, their action choices, and the possible payoffs. They are then asked to enact a role; for example, they are assigned the role of buyer or seller and asked

to make decisions (e.g., to buy or sell assets) that determine the amount they are paid for their participation. Payments are usually contingent on performance to motivate participants to deliberate more thoroughly on the task.

Experiments have provided an opportunity to test theoretical predictions; since experiments provide a sterile environment to control by confounding factors, researchers can use the method to test the predictive power of a theory—to reject or not reject hypotheses of predicted choices and reactions to changes in exogenous parameters (Ehmke and Shogren 2008).

Economic experiments have also turned out to be very useful in revealing and measuring some of these biases; they have allowed researchers to gather empirical data on how people make real decisions and how they respond to incentives and policies. They have also shed light on how people trade off individual and collective interest, how they cooperate and compete with others, and how such behavior varies among individuals and settings. For example, experiments have been applied to compare behavior among members of traditional societies from around the world and to test for different norms within different subgroups of society.

Experimental economics can make a considerable contribution to the design of environmental policies. Proposed policies and incentive mechanisms can be exposed to human decisionmakers in the laboratory, which may reveal any flaws or unanticipated outcomes. Experiments also allow alternative policies to be directly compared under controlled conditions, providing an empirical basis for policy selection. This can be done using a range of different parameters to indicate how each alternative might perform under various scenarios.

Economic experiments also provide a method for exploring how social preferences are affected by fairness concerns, or how different bargaining rules and protocols affect behavior and outcomes. Indeed, although environmental economics, like economics in general, is largely based on the normative criterion of economic efficiency, it is clear that many people have strong opinions about the perceived fairness of different environmental policies (Johansson-Stenman and Konow 2010). Behavioral economics have demonstrated that fairness arguments are clearly very prominent and decisive when it comes to the political viability of various instruments. It is crucial to integrate these insights properly into the environmental economics literature. Under some rules and settings, cooperation flourishes to the benefit of all concerned, while in other circumstances people revert to selfish behavior. Including these tendencies in research can contribute to better policies for managing environmental commons and public goods (Ostrom 2006).

A key feature of economic experiments is that participants are rewarded based on their decisions. This means that transactions in the laboratory have a real value. The way in which this is done is critical to the experiment design, since the induced values should match the economic system under investigation if realistic behavior is to be observed. On the other hand, although previous examples illustrate why behavioral economics could play a big role

in the way we think about environmental and resource economics, some researchers have questioned the robustness and relevance of some of the findings. In a sense, there will always be questions about whether the experimental environment is sufficiently rich to elicit behavior that is consistent with what would be expressed in natural economic environments with real economic commitments. Empirical tests and observation from naturally occurring situations are necessary to establish robustness and to identify those behavioral failures that are the most relevant for environmental policy to start the search for solutions. It is in many ways an exciting time as researchers are now finding they have a rich palette of methods ranging from theory to traditional empirical work but also from laboratory experiments to natural field experiments.

Supplemental Reading	
<i>Environmental Kuznets Curves (EKCs) and Resource Curse</i>	<i>Policy Reform</i>
Boschini et al. 2007	Atkinson and Stiglitz 1972, 1980
Dasgupta et al. 2002	Diamond and Mirrlees 1971a, 1971b
Dinda 2004	Feldstein 1976
Kijimaa et al. 2010	Hanemann 1995
Stern 2004	Murty and Ray 1989
<i>Environment and Development Economics 1997</i>	<i>Public Economics and Information</i>
Acemoglu et al. 2001	Arrow 1970
Adhikari 2005	Johansson 1997b
Grossman and Krueger 1995	Munk 1980
López 1994	Myles 1995 (especially chapter 9)
<i>Welfare Economics/Resource Economics</i>	Vickey 1969
Arrow 1951	Walters 1961
Debreu 1951	<i>Behavioral Economics</i>
Hartwick and Olewiler 1998	Brown and Hagen 2010
Mäler and Vincent 2001	Cardenas 2009
	Frey 1998
	Ostrom 2006
	Shogren and Taylor 2008

Notes

- 1 The relationship between these concepts is complex. *Development* is a broad concept that includes both economic growth and other (positive) societal changes, such as the addition of intangible value and maybe a more even distribution of income. *Growth* may mean simply an increase in GDP but ideally should be growth in true income (i.e., including various welfare-related aspects, such as the environment).

For the purposes of this discussion, the meaning approaches the more general concept of development, but the closer it gets, the harder it becomes to measure.

- 2 Similarly, the distribution of income, the degree of competition, the transparency of decisionmaking, the degree of corruption, and other variables are partly endogenous to economic development but can be influenced by policymaking and then, in turn, have a decisive impact on economic development.
- 3 Actually, more conditions are required, such as absence of indivisibilities and advantages to scale. More technically, all production and consumption sets must be convex, and all agents must have perfect foresight.
- 4 The complexities of society sometimes make it impossible to aggregate individual utilities to a social welfare function. If people cared only about individual income, it would work. However, if social concerns include altruism, if people have preferences concerning income distribution, or if welfare depends on relative rather than absolute levels of consumption, then the mere construction of aggregate welfare functions may not be feasible.
- 5 *Cost-effectiveness* means achieving the given goals at least cost. *Efficiency* includes the meaning of cost-effectiveness but also requires that the goals be set optimally with respect to welfare.
- 6 Not only does congestion cause delays and thus increased emissions per mile, but the rate of emissions itself is a function of speed and thereby of congestion (Johansson 1997a).
- 7 The optimization is now more complex. Utility (U) includes both private goods (x) and public goods (G) but also each individual's negative utility due to the externality (g) caused by others: $U(x, g_1, \dots, g_i, \dots, g_n, G)$. The Samuelson rule equates prices (transformation rates) to the sum of willingness to pay, including the sum of the (dis)utilities due to the externalities of congestion (Oakland 1972; also see Chapter 16).
- 8 The concept of rent as a payment for scarcity goes back to such classical economists as David Ricardo (1772–1823), who used “rent” to explain why land (when scarce) earns a form of income that is best measured by its marginal productivity.
- 9 However, the recent struggle between international drug companies and the government of South Africa over the right to import cheaper medicines for HIV (human immunodeficiency virus) shows that patents cannot always be followed. Whereas people understand that drug companies need to recoup their research and development costs, in the face of such a massive tragedy as the spread of AIDS (acquired immune deficiency syndrome), the main concern is access to inexpensive medicines. It becomes necessary to circumvent these patents to supply enough people with whatever therapy is available.

The Evolution of Rights

Property and ownership are formal expressions of the relationship between humans and the natural environment (Bromley 1991). Some economists would argue that the term *property* is a construct of the modern economy and that many people who live close to nature think of humans as “belonging to the land” or that the relationship is a divine one (e.g., Hanna and Jentoft 1996). In modern parlance, property consists of a bundle of rights. To properly analyze different forms of private or common property, the following typology is recommended (Ostrom and Schlager 1996)¹:

- 1 right of access: the right to use or enjoy the property for direct utility;
- 2 right of withdrawal: the right to use the property productively for profit;
- 3 right of management: the right to set up and modify rules for the use of a property;
- 4 right of exclusion: the right to exclude some users and set rules for access to a property; and
- 5 right of alienation: the right to sell, lease, or inherit property with the rights 1–4.

Any of these rights might be partial. For example, the right to inherit user rights but not sell those rights on a market is common. Other rights may be to move, change or adapt, and even destroy or dispose of property. According to Coase, these rights should be considered “factors of production” because they determine how much useful profit an owner can earn from a property (Coase 1960). Other important aspects of ownership are legal, cultural, and psychological—for example, the psychological satisfaction of ownership (as opposed to weaker rights to compensation) as embodied in the right to refuse to sell property rights (Rachlinski and Jordan 1998).

Bundles of rights may differ from one society to another. They typically have evolved through the interaction of neighbors and citizens living under specific circumstances. Common practice is to categorize the law broadly as private law or public law. Private law (or “common law”) refers to the old (particularly English) legal process of making and modifying the law on a case-by-case basis.

This law was made by the courts, and under the natural law doctrine, judges considered themselves “discoverers of the law.” (Popular notion was that a natural law governed the design of good relationships between people living together in a society, and these natural laws could be discovered through the observation of individual cases.) Public law is the more formalized process of creating statutes and constitutions through legislative bodies. It could, to varying extents, be considered the expression of popular will. The implied role of government within each kind of system is very different. Under public law, the legislature plays a dominant role in the definition of property rights, whereas under private law, the courts play the leading role.

Because adjacent properties inevitably share boundary lines and because noise, smell, energy, and materials easily flow across such boundaries, interdependencies in use usually exist. However boundary lines are drawn up and however property rights are allocated, some kinds of rights that an individual exercises on his or her property almost inevitably infringe on the exercise of rights by other individuals on their properties. For example, one person’s right to shoot a gun in any direction on his property limits a neighbor’s right to enjoy a bullet-free environment on her property. Similarly, one person’s choice of agricultural methods may have considerable effects on the aesthetics, air quality, water quality, and noise levels of adjacent neighbors. The right of one person to an unimpeded view is in conflict with the right of another person to erect high-rise buildings. The right to completely move or destroy property—such as, for instance, by removing all soil and turning a field into an open-pit quarry—is often restricted. The definition of property rights is closely related to the concept of externality, and the development of these rights over time is related to the transition from common property to private property. This process is often referred to as “enclosure of the commons.”

Property rights have evolved to gradually include more kinds of rights and property. An economist would consider this evolution as a response to human need to minimize uncertainty and transaction costs as well as a reflection of increased scarcity as population grows in a finite space. Notions of property rights may reasonably have stemmed from behavior relative to immediate possessions (e.g., food, shelter, and tools).² However, the formal history of property rights is closely related to the development of property rights to land. Therefore, the discussion of rights starts with private land, then moves to the somewhat more complicated rights (to water, minerals, and various forms of common pool resources). These discussions provide a basis for understanding the legal underpinnings of the rights to other natural and environmental resources, including less tangible objects such as biodiversity and clean air.

Real Property

The variations in the bundles of rights applicable to land property probably are greater than expected by those who are accustomed to any one set of laws. One example is the rights that people have on other people's land: rights to mining, water, and other resources automatically follow land ownership in some countries, whereas they are separate in others. In the United States, trespassing is highly illegal and asocial, whereas in Sweden, individuals have the right to walk; pick flowers, mushrooms, and wild berries; and camp (one night at a time) in private forests and other lands (not gardens or fields, however). Furthermore, these activities—and the associated rights of access and some partial rights of withdrawal—are not only legal but also accepted as normal. Similarly, in many developing countries, harvested fields are often used as a common property resource for grazing.

Differences in property law are a function of history. For example, the common law version of land law stems from the feudal relationships that developed after the Norman invasion of England in the eleventh century. Conquerors such as William I made way for a hierarchical system of property rights. The original inhabitants, the Saxons, were forced to give up land that was then appropriated by the Norman kings and their underlords. The king would grant use and occupation rights to lords in return for their loyalty. In turn, the lords could grant lesser rights to smaller parcels of land to tenants in exchange for some form of "payment," which could be an obligation to provide armed men or agricultural produce to the king or the lord, for instance. In modern parlance, these payments were forms of tax or rent for the right to use land. The feudal world was a hierarchical structure with the king at the top and many levels of lords and minor tenants below.

For the purposes of this discussion, the most central fact is that all the land essentially belonged to royalty. In fact, the word *real* in real estate refers to land owned by the king (*real* has the same roots as *regal* and *royal*, from the Latin *rex* or *regis* ["king"]). As the feudal structure developed, lords and their tenants inevitably bargained with each other. The lords wanted to extract as much labor, products, and services as possible from their land. The tenants wanted as many sticks as possible in their bundles of rights, especially the right to pass on their land to their children. This practice led to the process of land inheritance and a degree of certainty in land tenure.

Gradually, feudalism gave way to more democratic government. Representative governments took over from the king not only power but a whole system of government, property rights, and laws. By around 1500, many interests in land were recognized. The notion that the right to grant property essentially is vested in the king (or whatever government effectively rules the land) was widespread and can be illustrated by a famous American case, *Johnson v. McIntosh* (8 Wheaton U.S. 543, 1823), in which one party claimed ownership to a certain parcel of land in Illinois by virtue of purchase from an

Indian tribe before the Revolutionary War. The other party claimed ownership to the same parcel based on a later grant from the U.S. government. The first party sued based on the fact that its title was older. Justice Marshall argued that the dominant political power must have the ultimate authority to govern over property and issue land titles:

As the right of society to prescribe those rules by which property may be acquired and preserved is not, and cannot, be drawn into question; as the title to lands, especially, is, and must be admitted to depend entirely on the law of the nation in which they lie; it will be necessary, in pursuing this inquiry, to examine, not simply those principles of abstract justice ... which are admitted to regulate, in a great degree the rights of civilized nations ... but those principles which our own government has adopted in the particular case, and given us as the rule for our decision.

Justice Marshall therefore ruled that the title from the U.S. government was valid even though it was issued after the earlier sale.

Johnson v. McIntosh highlights an important fact: that a well-functioning property system requires certainty. Investors would be reluctant to invest money to build residential areas, set up businesses, and explore for natural resources if they were not somewhat certain that the law would enforce ownership rights. The *McIntosh* verdict hinged more on what was a good and practical institution for society than on what might be considered “fair”; according to many popular conceptions of “fairness” the first owner might have won by virtue of having older rights. This dilemma plagues the transitional countries of eastern Europe, particularly Russia, where many conflicting rights to the same land remain unresolved. The failure of government to properly establish the system of rights and liabilities has created an uncertainty that is detrimental to business. Government must somehow act as the ultimate “title” holder or at least guarantee property rights.

Still, the degree of government involvement and thus the extent of the private rights can vary extensively between countries, as illustrated in the example of trespassing on private property. Another major difference concerns a landowner’s rights to water and subterranean resources (such as oil or minerals) on a property. Some of the difference in economic dynamics between the United States and Mexico, for example, stems from historically diverging ideas about property rights to subterranean resources (Meyer and Sherman 1979). In Mexico, according to Spanish colonial tradition, all such rights are retained by the state (Mexico having been originally held by the Spanish crown), which gave landowners little incentive to mine. In the United States, mining rights are part of land rights, which was an essential element in creating a dynamic search for minerals and oil. On the whole, the U.S. system of property rights appears to have been crafted expressly to give more rights to individual landowners than to central government. This system presumably was created in reaction to the

overly centralized systems in Europe with its vestiges of feudalism. Even today, this distinction appears to be one of the underlying differences between culture and politics in the United States and Europe.

Outside these two regions, others have had government and ownership structures somewhat similar to feudalism, particularly Japan and China. Countries that have been colonized typically have layers of legal systems with different perceptions and definitions of property rights. In policy design, considering the structure of all kinds of property rights is crucial, and in many developing countries, common or communal property is particularly important. Intermediate forms of property that have private user rights but are owned by the local community also are fairly common. In Ethiopia, in fact, many forms of traditional land ownership were structured in this way (Alemu 1999).

In northern Ethiopia, the *rist* tenure system has predominated. Traditionally, everyone in a village had a right to land to support his or her family. People who left the village would temporarily lose their rights to use the fields but could reclaim them on their return. Today, these rights have been modified. The council (known as the *Derg*) that seized power in a 1974 socialist revolution nationalized all land. Even though the *Derg* collapsed in 1991, the new Ethiopian Constitution from 1994 still affirms that all land and natural resources belong to the state. However, the new regime has strengthened customary user rights to land, and in many parts of the country, the old traditions continue to dominate everyday life. Even distant descendants who can prove their lineage to village elders can come back and claim land rights. Because some families have many children and some have none, making the average holding per family uneven, and because villagers have the right to reclaim land rights, even after several generations, there is a tradition of periodic land reallocation. These reallocations are different from the modern notion of individual property rights, and risk of reallocation might be expected to diminish incentives for investment significantly. However, reallocations solve the problem of combining equity with hereditary rights as perceived, and they appear to have been fairly common in many African countries. And surprisingly, African systems of land tenure with reallocation generally do not have a negative influence on productivity (Place and Hazell 1993; Migot-Adholla et al. 1993).³

The system of land reallocation had significant repercussions on investments during the *Derg* regime (the so-called socialist era), when reallocations appear to have been frequent, brutal, and made partly based on allegiance to the ruling party. The original system was not necessarily illogical, but in a new historic context, frequent and erratic reallocations created a general lack of tenure security. Now that technology and investments have become so important in financing modern agriculture and increasing productivity, the absence of tenure security and thus of investment incentives has had a significant effect on economic growth (Alemu 1999).

Peasant associations are ensnared in a seemingly impossible trap because they are responsible for conflicting tasks: managing the common pool resources

of the village as well as allocating “individual” land lots as equitably as possible. Compounding this difficulty, young adults increasingly want to set up their own households, and many demobilized soldiers and other settlers are returning to their villages. Because individual plots are already small, taking land from existing community members is extremely difficult, so the village association is essentially forced to use common pool resources, which thus effectively become privatized (Kebede 2001).

In Papua New Guinea, some successful examples of communal land rights do not allow for free access. Families have individual and indefinite farming rights, but the rights of sale belong to the clan. Experience with the privatization of land has been bad. Rather than clarify rights, privatization created uncertainty and confusion, because the concepts of private land ownership are not well integrated into local culture. Many lands that had been converted to freeholds were later disputed and, on many occasions, returned to communal ownership (Panayotou 1993).

Holden et al. (2008) is the first systematic attempt to address emerging land markets and their implications for poverty, equity, and efficiency across a number of African countries. While land markets and especially informal markets have been rapidly emerging in densely populated parts of Africa—often to the benefit of the poor—their functions remain imperfect, because of tenure insecurity and the fragmentation of agricultural land. Rigorous quantitative analysis designed to account for the role of land markets in national land policies demonstrated that, all too often, land policies have been extreme—either prohibiting all land transactions or giving unrestricted freehold rights to a small elite at the expense of the poor. From the long experience in Asia, it is known that such policies are detrimental to both production efficiency and equity of land use. Holden et al. (2008) argue that future policies in Africa should work with the markets. Regulations and institutions should be imposed only with careful testing to make sure they have the intended effects.

Common Property Resources

In the history of property rights, more and more objects and attributes of objects gradually become defined as “property.” Beyond the realms of personal belongings (e.g., food, clothing, tools, and housing), land, and mineral resources remain ecosystem resources: waterways, forests, and wild animals. Roman law called these kinds of property *res communes* (“things owned in common”). *Res communes* could not be owned in their natural state by anyone. They were thought of as belonging to the “public” as a whole: “By natural law these things are common to all: air, running water, the sea and as a consequence, the shores of the sea” (from *Institutes of Justinian*, quoted by Wiel [1918], who adds “fish, wild animals and game, the light and heat of the sun, and the like”). This sentiment is still often expressed, partly in

opposition to one prevalent aspect of economic development, the “enclosure of the commons.”

In England as elsewhere, the commons historically were areas of lands that were less productive and therefore did not justify the transaction costs (including draining and fencing) necessary for agriculture. These lands provided important ecological, recreational, and other services to everyone; they were particularly important to the poor as sources of fodder and pasture (for livestock), firewood, water, fishing, and many other uses. With increased population density and changes in social structure, technology, and other factors, more and more of the commons were privatized or “enclosed” (historically, private land had to be enclosed by some form of fence or boundary). The political struggles over the enclosure of the commons lasted for centuries in England and were sometimes fierce. In one respect, they can be seen as a process of modernization that increased productivity, but in another respect, they were a struggle in which the poor attempted to protect their rights to pasture against the landlords’ desire to extend private holdings.

Today, similar struggles still take place all over the world—especially in poor countries—and the distribution of land between kinds of property (e.g., private, state, and common property) has significant importance not only for the ecosystems as such but also for the distribution of wealth and welfare in society. A study of 75 villages in India found that 30%–50% of common pool resource areas had been lost between 1950 and 1982. In 1950, 70 of those 75 villages had rules for managing common pool resources, but in 1982, only 8 of them still did. Similarly, whereas 55 villages used to collect (formal or informal) taxes to maintain the common pool resources, all of them had stopped the collection by 1982 (Jodha 1992).

In addition, new “items”—biodiversity, radio frequencies, the high seas, the Arctic regions, outer space, and genetic and other information—are being made into “objects” of property rights in a way that is structurally reminiscent of the enclosure of the commons. A striking example is the “enclosure” of the seas implied by U.N. Third Conference on the Law of the Sea in 1982. A main result of the conference was an extension of the exclusive economic zone to 200 miles, which implies jurisdiction by coastal states over fishing in this area and an appropriation of most of the value from the oceans by the coastal states of the world.

Another pertinent example comes from Monsanto Company, a producer of agricultural products. Because Monsanto patents the genetic code of the seeds it genetically modifies, farmers that use proprietary seeds must sign a contract that prohibits seed saving (i.e., using seeds from harvested crops to plant future crops). Planting harvested seeds is well established in agriculture and could in fact be considered the foundation of human civilization. Yet biotechnology is now trying to outlaw this practice.

Some ecologists see this and similar forms of enclosure as the last frontier in a losing battle between the ecosystem and humanity on one side and

capitalism on the other. Others take the opposite view, that the existence of externalities and the “tragedy of the commons” depends on the lack of appropriate definitions for property rights. With better technology—both legal and physical (for fencing, supervision, detection, communication, and enforcement)—more detailed and precise property rights could be defined, which, according to this view, would not only enhance efficiency but also minimize the extent of negative externalities and degradation to natural resources. Even in the instances where this is true, the poor might be at a disadvantage. By definition, poor people have no capital and few productive assets. For this reason, access to commons or even to degraded open-access areas may constitute a significant, even crucial, contribution to their welfare. Thus, in some instances, enclosure can raise overall productivity while further impoverishing the people in the poorest segment of society, who benefited from the natural resources in the area of concern. Today we are in the midst of the biggest “enclosure” yet: that of the atmosphere (or its content of “greenhouse” gases). As we will argue elsewhere in this book, this is a struggle that could go on for many decades—except for the fact that we need to find a solution fast.

Although a historic trend toward enclosure seems to be apparent, common property is not necessarily an inferior kind of property. In several respects, a well-designed, well-functioning common property resource (CPR) is like private property (Ostrom and Schlager 1996; Murty 1994). Notably, CPRs imply the exclusion of outsiders and thereby make it possible to avoid the “tragedy of open access.”⁴ In some ecological and social contexts (when the costs of protecting private property are high or when the yields are low and very variable), a CPR may simply have lower transaction and other costs and thus be more efficient than private property.

Water Law

Water is the ecosystem resource that, after land, probably has the most fully developed law. It also has many attributes that help us understand property rights for other ecosystems, such as mobility, variability, and unpredictability. Two fundamental doctrines in water law are *riparian* and *prior appropriation*.

The riparian doctrine, the more common of the two concepts, developed in Europe and has its roots in Roman law (Teclaff 1985).⁵ This influence is especially clear in the French Napoleonic Code. The essence of the riparian water doctrine is that the right to use water depends on access to the water through ownership of adjacent land. The owner of property adjacent to a river or stream has certain rights to the flow of that water by virtue of land ownership. Irrigation, a chief concern in developing French law, is incorporated in the riparian concept:

He whose property borders on a running water may serve himself from it in its passage for the watering of his property. He whose estate such water crosses is at liberty to use it within the space which it crosses, but on condition of restoring it, at its departure from his land, to its ordinary course. (Wiel 1918, 258)

The principle of extending land ownership to adjacent ecosystems or geophysical objects also arises in the allocation of mineral rights and in the appropriation of the seas by coastal nations. An early American Supreme Court Justice borrowed from the civil law system (especially the Napoleonic Code) to develop the modern U.S. riparian doctrine (Wiel 1918, 245–247).

The prior appropriation doctrine of “first come, first served” began in California gold-mining camps in the mid-1800s and was developed most fully in the United States, mostly in the West. This doctrine gives the first rights to the first user in time; in other words, it protects the first person who uses a stream’s water from subsequent appropriations; the first person to put water to a beneficial use has a right superior to that of subsequent appropriators.⁶ Subsequent appropriators are allowed to use water only to the extent that they do not impede the use of those who have prior rights. The doctrine establishes a priority system in which the senior appropriators can “call” the river in dry years to prevent appropriations by water users with more junior rights until the senior users have received their full shares of water. Unlike the riparian doctrine, it is not necessary to own land that is adjacent to a river or other body of water to obtain a water right. All that is needed is to be first and to use the water beneficially.

Another category of rights is the right of state governments. Many states declare, often in their constitutions, that all waters within their boundaries are the property of the state. The state may then allocate water rights through an administrative system that is commonly divided between watersheds. In the United States, for example, a person who wants to use water submits an application to the appropriate agency, stating the source, method of diversion, and intended use of the water. Normally, there are no (or low) charges for the water. The actual water right is not vested until the water is diverted and put to a beneficial use—tying in with the prior appropriation concept. After the water right is recognized, the state issues a title for a certain quantity of water for a given purpose—say, irrigation. The quality of the water right depends on the priority date (i.e., the date of first beneficial use or the date of application). In the dry summer months, some owners may get no water at all because the users who have older water rights are entitled to take their water first. A water engineer will shut down a water supply head gate (drinking water excepted) to first supply senior water rights. It is perhaps typical that this prior appropriation system developed in a region where water is (or at least was at the time) much more scarce or valuable than land. No strong traditional landowners were around to speak for riparian-style rights.

Spanish water law has had various influences and shows the interplay among the various doctrines. One of the earliest written laws from the seventh century, the *Fuero Juzgo*, had elements of both Hispano-Roman and Germanic law. The common and usufruct rather than private nature of water was emphasized: “No one shall for his own private benefit, and against the interests of the community, obstruct any stream of importance; that is to say, one in which salmon and other sea-fish enter, or into which nets may be cast, or vessels may come for the purpose of commerce” (*Visigothian Water Code*, Law 29 of Book 8, Title 4, quoted in Wiel 1918).

Moorish rule (from the early 700s to 1492) later imparted the practice of communal management, particularly in Valencia. The Tribunal of Waters had considerable elements of CPR management, and land typically was not sold without its associated water rights (Ostrom 1990; see also Chapter 23). By the thirteenth century, Roman law again became important, as Castile kings such as Alfonso X sought to centralize water law in order to be able to grant water rights and possession of canals to landowners. The dilemma was that making water common had to include rights to cross private lands with canals and pipes, but it also made more irrigation possible.

Roman law was codified around 1260, establishing once again that water was *res communes*, not susceptible to ownership. All rivers were public property, except that feudal systems influenced Spanish water law, giving property rights to kings and nobles. The effect was moderated by the communities—*pueblos*—that protected water sources for their members. Throughout the centuries, the struggle between legal principles and among groups of users continued. Gradually, feudal influences were minimized, and in 1831, A Broad Compilation of Water Law declared that rivers that follow their course to the sea “cannot be private property” (quoted in Dobkins 1959, 78–79). This declaration was necessary partly because of the importance of navigation. However, for nonnavigable streams, irrigation rights were liberalized.

Not surprisingly, Spanish and French laws heavily influenced laws in their colonies, especially in the Americas. The system of local control appears to have been the aspect that was most easily transferred.

Lessons for Environmental Externalities and Commons

The riparian concept is founded on (land) property rights and also has something of an ecosystem approach, whereas prior appropriation emphasizes beneficial use and is “first come, first served.” The oil industry is also full of interesting lessons. In the early development of the U.S. oil industry the “law of capture” came to be accepted as the dominant legal principle. This principle says that the oil (or gas) becomes the property of whoever “captures it” by drilling on land he or she owns or has legal access to even if that oil or gas migrated from under adjoining lands belonging to someone else. It has some

element of similarity with the riparian concept since it is tied to land tenure but also to rights over wildlife (since wildlife belongs to no one until it is captured, and the right to shoot may or may not be tied to land ownership). It has some similarity with prior appropriation in that it protects the interests of producers—but there is an important distinction. If A drills a well at the boundary of his property facing B who already has a well, this will often reduce production for B and might be judged as a negative externality—or even a theft of the “prior rights” of A (as, for instance, in Californian water law). Not so in the case of the oil industry, as discussed in Daintith (2010), a detailed account of the development of the law of capture, a rule that clearly encouraged a vast over-capitalization and race for resources in the early oil industry.

One way to think about new rights to various ecosystem and environmental resources is by extrapolation from the land, water, and mineral examples. In this section, we briefly comment on cases concerning air pollution, noise, nuisance, and risks as well as unclaimed property, such as some beaches, shallow fishing grounds, and virgin forest.

If an individual or a firm can acquire water rights by prior appropriation, then it might be argued that a factory that has been openly emitting smoke for several years for productive purposes should have rights to the assimilative capacity of the atmosphere according to the principle of prior appropriation. Some people consider this argument supportive of the allocation of emissions permits by grandfathering, that is, in proportion to historic use.

The courts often perceive their role as one of balancing legal principles, and in so doing, they may make reference to what is a “reasonable use” or to which arrangement gives the greatest total benefit to society. These concepts are illustrated by the erosion of the principle of “nuisance” or “trespass.” For example, in the case of *Hay v. Cohoes Co.*, Hay uses dynamite to build a canal. Consequently, rocks fly into an adjacent property. The Court of Appeals (New York, 1848) held that “(a) man may prosecute such business as he chooses upon his premises, but he cannot erect a nuisance or annoyance of the adjoining proprietor, even for the purpose of a lawful trade.” At the time, the neighbor’s right to a safe environment dominated. However, a few decades later, in *Booth v. Rome, Watertown & Ogdensburg Terminal Railroad* (1893), the blaster was granted the right. Even though the plaintiff’s house had been hit by rocks, the New York Court of Appeals held that the blasting did not create liability if carried out with “reasonable care.” The court explicitly incorporated a utilitarian consideration: a city could never be built without dynamite (Green 1997).

According to the Coasian tradition, the externalities imposed on a victim by a polluter and the costs imposed on a polluter by a victim are completely symmetric (Coase 1960). In the case of *Sturges v. Bridgman*, for example, a confectioner has been making candies in the same locality for more than 60 years. A physician builds a practice directly adjacent to the confectioner and then discovers that the pounding noise of the confectioner’s mortar and pestle

makes him unable to use a stethoscope and even affects his ability to think. Even though the doctor is the newcomer and reasonably knows about the confectioner's business, he is awarded an injunction, and the confectioner has to cease his business.

In some cases, externalities are completely symmetric (e.g., in congestion, overgrazing, and overfishing). Many observers might argue that there is unidirectionality in the case of *Sturges v. Bridgman*; after all, only the confectioner makes noise. However, "imposing silence" creates just as much of an externality as making noise: either the confectioner works and the doctor cannot, or the doctor works and the confectioner cannot (Coase 1960). One might still find this verdict odd because the confectioner has prior rights, but the court followed what it perceived to be the maximum social utility.

Coase emphasizes that marginal analysis can lead to the wrong conclusion. An aggregate profit function for the confectioner and the physician might be nonconvex and have several maxima. A little more noise might increase profit for the confectioner without affecting the doctor, who already cannot work at all because of the noise. The absolute social optimum might be no noise and no confectioner. Thus, both total and marginal analyses are needed. Also, if there were no transaction costs or wealth effects due to the reallocation of rights, then the outcome would be the same if the doctor had paid the confectioner to be quiet or if the confectioner had paid the doctor as compensation for the noise.

However, if there were transaction costs, then the outcomes might be different. Coase believes that in such cases, society should allocate rights so that the maximum social utility is achieved; a property allocation that is intended to maximize social utility would depend on the details of a particular case.⁷

In traditional (i.e., sparsely populated and resource-rich) societies, relatively few rights are formally defined; in modern societies, more and more resource rights are narrowly defined. The institutions of property appear to have evolved in a way that is roughly proportional to the economic advantage (scarcity rent) involved. The exploitation of natural resources can be seen as a race for property rights, much like the great move to the U.S. West in the 1800s, the California gold rush (1848–1855), or the current colonization of the Amazon and other developing-world ecosystems.

When concepts of property rights clash—particularly when modern codified concepts take over from the more informal, culturally rooted, usufruct systems—the result can be an uncertainty that leads to a wasteful abuse of resources and, in many cases, to solutions that are perceived as unfair by the original users. For example, in the traditional island society of Mafia (close to Zanzibar), ownership rights are defined not to land (the so-called "coral rag," because it has low productivity) but to fishing sites and to coconut trees, which are valuable and belong to whoever planted them. When hotels want to buy water, land, beach rights, and so forth, negotiations tend to be uneven; hotel owners often have been successful in securing "titles" that local residents do not understand or respect (see Chapter 28).⁸

Many nation-states have taken it upon themselves to nationalize ecosystem properties—making the lands government property instead of common pool resources—allegedly to protect those resources from degradation. The consequences usually are swift and harsh, as in the central hills of Nepal, where nationalizing the forests in 1957 quickly led to deforestation because villagers no longer perceived themselves as the owners or beneficiaries of the forests. During the past few decades, forest management has reverted back to village communities, and reforestation has been fairly rapid.

Although government plays a vital role as an ultimate guarantee of property rights per se, it has often failed abysmally as a direct owner and manager of natural resources, partly because of the general factors behind policy failure. Several factors suggest why governments fail to turn claims to ownership into effective control and management (Panayotou and Ashton 1992):

- enormous size of areas nationalized (in many countries, more than one half of total area),
- fast rate of this transfer from local control to national control,
- failure to recognize and respect local customary rights,
- limited budget and administrative capacity to manage the natural resources,
- increasing population pressure, and
- failure of rural development to provide alternative employment.

At the other end of the scale, it is sometimes asserted that all environmental problems could be solved if property rights to all relevant resources (including waterways, the atmosphere, and various attributes of land—biodiversity, minerals, water, radio frequencies, and air space) were to be separately (and privately) allocated. As soon as an externality was created between two rights—say, one person’s right to build a second story on his house and his neighbor’s right to enjoy an unobstructed view—new “rights” to views or height would be created to remove the externality by converting it into a market transaction.

Private allocation of property rights to natural resources may be useful in some instances but does not always provide appropriate solutions. The more rights are divided, the more boundaries exist between them and the greater the likelihood of creating new externalities. After all, externalities are essentially physical connections between attributes of different properties—particularly when underlying production functions are strongly nonlinear, like when there are strong synergistic effects in the environment. For example, assigning responsibility, liability, or rights becomes increasingly difficult in a context where hazards are caused as the joint effect of pollution from several separate but neighboring entities.

Sometimes it is preferable to assign (or maintain) some form of collective property rights. Another aspect of privatization that is important for social welfare is that poor people have little property other than common

property resources or open-access properties. In this perspective, privatization or enclosure is a disenfranchisement of the poor and likely to exacerbate their poverty and thus be violently opposed. One of the challenges for environmental economists is to design systems for property rights that maximize welfare (taking into account the costs of externalities and the “overgrazing” that results when exclusion is ineffective, as well as the distributional and cultural aspects). During the past few decades, countries have begun to come together to create mechanisms to build up an international governance of the global commons (for example, the Kyoto Protocol, or the Montreal Protocol, in which cost distribution is a particularly sensitive issue; see Chapter 8).

Acknowledgments

This chapter builds on excellent research assistance by John McClanahan.

Supplemental Reading

Baca 1993	Platteau 2000
Cohen 1954	Stone 1995
Daintith 2010	Toulmin and Quan 2000
Oxman 1997	Wolf 2001
Pigou 1932	

Notes

- 1 Using the same notation (Ostrom and Schlager 1996), the salmon and herring fishermen of Alaska are “authorized users” with rights 1 and 2 because they can fish only with permission from a commission. The cod trap fishermen of Newfoundland are “proprietors” but not “owners” because they have rights 1–4 but not 5.
- 2 Compare this behavior with that of a fox or a wolf defending its “property”: the intensity of its aggression is a function of how close it is to the center of its territory (Lorenz 1966).
- 3 Thanks to Hans Binswanger for discussions on this point.
- 4 This is a conscious paraphrase of Garrett Hardin’s famous (but somewhat mistaken) “tragedy of the commons,” the point being precisely that commons may be well managed, whereas open-access resources may not be.
- 5 This section draws heavily on Wiel 1914 and 1918 and Dobkins 1959.
- 6 *Beneficial use* appears to stem from a deeply rooted feeling that owners have to use their property to defend their rights. It may apply even to such secure forms of property as a house: in some societies (such as England), if a house is abandoned and squatters move in, the owner may lose partial rights to it. One perverse example of beneficial use took place in the Amazon, where forest clearing was a criterion of beneficial use, and deforestation thus became a means of securing titles (Binswanger 1991) (see Chapter 27).

- 7 Ownership rights also encompass cultural and psychological aspects. For example, in *Boomer v. Atlantic Cement*, eight neighboring property owners were granted compensation but not injunction (not “property” rights—i.e., the right to stop the plant’s pollution). Despite compensation, the plaintiffs continued to litigate. A Coasian analysis would be that they were merely trying to improve their bargaining position to get even more compensation; however, modern analysts conclude that there is some psychological attachment to ownership, which questions a central element of Coasian thought (Rachlinski and Jordan 1998).
- 8 The diversity of laws, customs, and attitudes with respect to ecosystems in different countries cannot be addressed here, but many non-Western civilizations have interpreted their relationship with nature in divine rather than material, legal, or economic terms. In the ancient Indian religious literature (Vedas, Upanishads, smriti, and dharmas), for example, the sun, air, water, earth, and forests are personified as gods and goddesses (Sankar 1998).

PART II

Review of Policy Instruments

In this section, we present the main categories of policy instruments used for environmental and natural resources policy as well as short descriptions of how each instrument works. The purpose is to provide background and context for the chapters on criteria and policy selection in Part III.

Policy does not function in a vacuum; it is heavily dependent on the overall policy environment. If the economy is not competitive and if the bureaucracies are not honest, well-informed, and sufficiently well funded to carry out their responsibilities, then no policy instrument will work perfectly—although some will work better than others.

Information plays a special role in policymaking, and in fact, information provision can be considered an instrument in its own right. On a general level, all policy depends on information; that is, policymakers must understand the technology and ecology of the issues under consideration. Before the ozone-depleting action of chlorofluorocarbons (CFCs) in the stratosphere was understood, for example, no CFC-regulating policy existed. A general policy based on the precautionary principle could have been formulated, but before the scientific connections are suspected, serious restrictions are usually considered overly cautious.

Much discussion on policy instruments is conducted as if there were only two instruments: standards and taxes; however, many instruments exist, with

many characteristics. Policy instruments are often classified as “market based” versus “command and control,” but this classification is poor. Markets involve prices and quantities, regulations often are backed by economic sanctions, and even economic theory suggests that quantitative instruments such as standards, emissions targets, or permits may be optimal in many cases.

Some political scientists insist that there are only three basic categories of policy instruments, aptly nicknamed “carrots, sticks, and sermons” to symbolize economic incentives, legal instruments, and informative instruments, respectively (Bemelmans-Videc et al. 1998). This classification can be further differentiated into physical, organizational, legal, economic, and informative categories (Lundqvist 2000). No single taxonomy is necessarily preferable, but each can be useful in a different context.

Many policy matrices have been proposed as organizing principles for the systematic collection and comparison of policy experience. One useful typology (based on World Bank 1997) for organizing the rich diversity of actual experiences in the field is presented in Table II-1 and Table II-2. The policy instruments are divided into four categories: using markets, creating markets, environmental regulations, and engaging the public. The various kinds of policy instruments listed are applicable in the area of natural resource management (water, fisheries, land, forests, agriculture, biodiversity, and minerals) or in pollution control (air, water, and solid and hazardous wastes).

The first category of instruments, “using markets,” includes subsidy reduction; environmental charges on emissions, inputs, or products; user charges (taxes or fees) (see Chapter 5), performance bonds, deposit–refund systems, and targeted subsidies. It also includes instruments such as refunded emissions payments and subsidized credits (see Chapter 7).

The next category of instruments, “creating markets,” consists of mechanisms for delineating rights. The most fundamental of these mechanisms has particular relevance in developing and transitional economies: the creation of private property rights for land and other natural resources. A mechanism that

TABLE II-1 Classification of Instruments in the Policy Matrix

<i>Using markets</i>	<i>Creating markets</i>	<i>Environmental regulations</i>	<i>Engaging the public</i>
Subsidy reduction	Property rights and decentralization	Standards	Public participation
Environmental taxes and charges	Tradable permits and rights	Bans	Information disclosure
User charges	International offset systems	Permits and quotas	
Deposit–refund systems		Zoning	
Targeted subsidies		Liability	

Source: Adapted from World Bank 1997.

is relevant at the local level is common property resource management. Special kinds of property rights in environmental or natural resource management are emissions permits and catch permits (see Chapter 6). In an international context, such mechanisms are often referred to as “international offset systems.”

The category “environmental regulations” includes standards, bans, (nontradable) permits or quotas, and regulations that concern the temporal or spatial extent of an activity (zoning) (see Chapter 4). Licenses and liability rules also belong in this category, connecting it to a large area of lawmaking and to the politics of enforcement. Such instruments as liability bonds, performance bonds, and (more generally) enforcement policies and penalties are all part of the instrument arsenal.

The last category, “engaging the public,” includes such mechanisms as information disclosure, labeling, and community participation in environmental or natural resources management (see Chapter 8). Dialogue and collaboration among the environmental protection agency, the public, and polluters may lead to voluntary agreements, which have become a fairly popular instrument recently.

Not included in Table II-1 but still potentially important in various contexts are four mechanisms: direct provision of environmental services (such as municipal waste disposal); international agreements (which are only a policy at the multinational level); environmental auditing and certification, which is mainly a policy instrument at the firm level (often used in conjunction with labeling and information provision); and macro policies in general (all fiscal, monetary, and trade policies have implications for the whole economy and thereby for the environment).

Because of the wealth of information available, policy matrices easily can become unwieldy. Table II-2 is a simplified example of a matrix that illustrates the policy matrix approach and simultaneously provides a “map” of this book by listing most of the instruments discussed.

TABLE II-2 The Policy Matrix: Instruments and Sample Applications

<i>Policy instrument</i>	<i>Natural resource management</i>	<i>Pollution control</i>
	<i>Water, fisheries, agriculture, forestry, minerals, and biodiversity</i>	<i>Air pollution, water pollution, solid waste, and hazardous waste</i>
Direct provision (4)	Provision of parks (28)	Waste management (24)
Detailed regulation (4)	Zoning (26, 28) Regulation of fishing (e.g., dates and equipment) (25) Bans on ivory trade to protect biodiversity (28)	Catalytic converters, traffic regulations, etc. (19) Ban on chemicals (20)
Flexible regulation (6)	Water quality standards (23)	Fuel quality (18) CAFE (18)

continued

TABLE II-2 (Cont.)

<i>Policy instrument</i>	<i>Natural resource management</i>	<i>Pollution control</i>
	<i>Water, fisheries, agriculture, forestry, minerals, and biodiversity</i>	<i>Air pollution, water pollution, solid waste, and hazardous waste</i>
Tradable quotas or rights (6)	Individually tradable fishing quotas (25) Transferable rights for land development, forestry, or agriculture	Emissions permits (20)
Taxes, fees, or charges (5)	Water tariffs (23) Park fees (29) Fishing licenses (25) Stumpage fees (27)	Waste fees (24) Congestion (road) pricing (2, 16) Gas taxes (17) Industrial pollution fees (20)
Subsidies and subsidy reduction (7)	Water (23) Fisheries (25) Reduced agricultural subsidies (26)	Energy taxes (20) Reduced energy subsidies (20)
Deposit-refund schemes (7)	Reforestation deposits or performance bonds in forestry (27)	Waste management (24) Sulfur, used vehicles (16) Vehicle inspection (19)
Refunded emissions payments (7)		NO _x abatement in Sweden (20)
Creation of property rights (8)	Private national parks (28) Property rights and deforestation (27)	
Common property resources (8)	CPR management (26, 27, 28)	
Legal mechanisms, liability (8)	Liability bonds for mining or hazardous waste (24)	
Voluntary agreements (8)	Forest products (27)	Toxic chemicals (6, 24)
Information provision, labels (8)	Labeling of food, forest products (26, 27)	PROPER and other labeling schemes (24, 25, 27)
International treaties (14)	International treaties for protection of ozone layer, seas, climate, etc. (14)	
Macroeconomic policies (2)	Environmental effects of policy reform and economic policy in general (2)	

Notes: Numbers in parentheses indicate chapters in this book. Columns have been “collapsed” to save space. Chapter numbers in the second column indicate where the general explanation is given; other chapter numbers refer to empirical examples. Some examples have no chapter numbers because they are not discussed at any length in this book. These and other new examples of policy instruments may be found at two World Bank websites: Environmental Economics and Indicators (www.worldbank.org/environmentaleconomics) and New Ideas in Pollution Regulation (www.worldbank.org/nipr/commun.htm).

Direct Regulation of the Environment

This chapter starts with a short section on how to judge the optimality of policy instruments. Then, we discuss three different kinds of physical instruments in turn: *direct provision by the state of public goods*, the *direct regulation of technology*, and the *regulation of performance*. Although the last two categories are sometimes grouped together, they differ considerably in the freedom given to individual firms.

Optimality and Policy Instruments

The policymaker's situation is illustrated simply in Figure 4-1, which shows the results of quantity-type policies to cut back pollution from current emissions (E_0) to an optimal level of emissions (E^*), and price-type policies, which impose a tax or fee (T^*) on polluters, thus motivating them to cut back pollution to E^* . The marginal cost of abatement (i.e., the extra cost incurred in achieving a lower level of pollution) is derived mainly from the costs of clean production methods. In this simple model, both kinds of policies lead to the same result. In reality, the decisionmaker has many more instruments to choose from, but there is generally less certainty concerning their effects.

The basic purpose of policy instruments is to correct for market (and institutional) failures such as externalities. In Figure 2-3 (Chapter 2), it is assumed that rates of pollution are fixed, and thus, emissions can be limited to E^* only by limiting production to quantity Q^* . Two ways of achieving these limitations are by regulating emissions or production quantity, or by internalizing the externalities by a Pigovian tax (T) equal to the marginal cost of the environmental damage done by dirty production (MCE). If emissions factors are assumed to be fixed, then an emissions tax can be replaced by a product tax, but such an assumption is restrictive—the amount of emissions generally cannot be considered proportional to the amount of product produced. Many environmental cases are “cleaned up” not by reducing output but through abatement, and an important function of policy instruments is to encourage such abatement efforts.

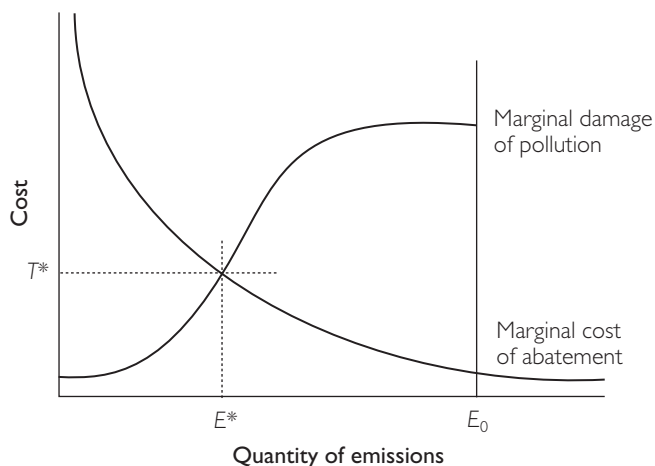


FIGURE 4-1 Optimal Emissions Reduction

Note: T^* = optimal tax or fee; E^* = optimal emissions level; E_0 = current or initial emissions.

To analyze abatement, the simple model is inadequate; the possibility of cleaner modes of production must be introduced, as in Figure 4-2. If there are two modes of production, one dirty with a marginal cost of MC_D and one completely clean mode of production with a marginal cost of clean production (MC_C), and if $MC_C < T + MC_D$, then the product price will not rise to P^* but only to P^{**} , and the decrease in output will also be smaller (see Chapter 11). The main influence on price is the adoption of clean technology. The clean technology might also have smaller external effects that lead to residual tax payments, but this possibility is ignored for the sake of clarity.

This section of the book is dedicated to describing the function of policy instruments. The standard against which to compare other instruments is provided by considering the maximization of welfare (Box 4-1), illustrated in the simplest possible case of a flow pollutant (for which there is no accumulation and thus no intertemporal aspect) and only one damage function—which means that there is only one pollutant and “perfect mixing” (i.e., no spatial dimension is taken into account). Box 4-1 shows two important requirements for optimality in this model. First, the marginal abatement costs in production should be set equal to the marginal benefit of the abatements as measured by a reduction in environmental damage. Second, the price of the product produced should reflect not only conventional production costs but also the scarcity rent that corresponds to the environmental damage that is caused despite the abatement costs incurred. This equation is essentially a formulation of the “polluter pays” principle. Some industrialists have complained that they are forced to pay twice—once for abatement, and once for pollution damage—but this scenario is analogous to the way the market sets the prices of other scarce items, such as water. For example, even though

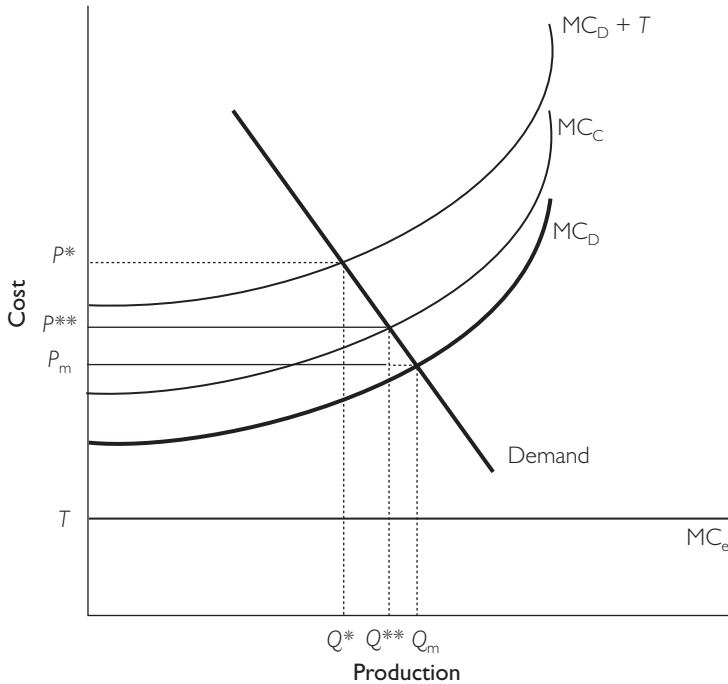


FIGURE 4-2 Pigovian Externality Taxes and Abatement

Notes: P^* = optimal product price in the absence of a clean production technology; P^{**} = optimal product price with clean production technology; P_m = market product price in the absence of policy; T = environmental tax level; MC = marginal cost of clean (C) or dirty (D) production; $MC_D + T$ = social cost of dirty production; MC_e = marginal environmental damage caused by dirty production; Q^* = optimal quantity in the absence of a clean production technology; Q^{**} = optimal quantity with clean production technology; Q_m = market determined quantity in the absence of any policy.

a firm purchases equipment that decreases overall water usage, the firm still is obliged to pay for the water consumed. Thus, a firm that releases pollutants to the environment should pay for the assimilative action of the atmosphere.

If the assumption of perfect mixing is relaxed and each source is assumed to have a specific transfer coefficient (perhaps inversely related to the distance from the source to a particular receptor point, such as a town), then each source will contribute a different amount to each of several receptor points. Pollution levels usually vary between these receptor points, and damage functions must be modeled individually for each of them.

Two conditions are modified to reflect the sum of damages done by each individual source to all the receptor points. Emissions from source i (e_i) thus cause damage at receptor j of $D_{ji}(e_i z_{ij})$; total damage at this receptor point will be $D'_{ji}(e_i z_{ij})$. Total damage at all receptor points will be $\sum_j D'_{ji}(e_i z_{ij})$ giving the following alternative conditions:

BOX 4-1 Optimal Abatement

Consider a social planner who maximizes welfare (W), a function of net profits minus environmental damage of emissions, both of which depend on output (q) and abatement (a). Emissions are assumed to be perfectly mixed so that the damage (D) is equal to $D(E)$ is equal to $D(\sum e_i)$, where E is total emissions and e_i is the emissions of firm i :

$$W = \sum_i [Pq_i - c_i(q_i, a_i)] - D[\sum_i e_i(q_i, a_i)] \quad (4-1)$$

where P is product price and c_i is production costs for firm i . The necessary first-order conditions (dropping index i) are

$$P = c'_q + D'e'_q \quad (4-2)$$

$$c'_a = -D'e'_a \quad (4-3)$$

D' , c' , and e' are derivatives of the damage, cost, and emissions factors, respectively. Subscripts indicate derivatives with respect to variables q and a . The necessary conditions imply that optimal price should internalize damage costs, $D'e'_q$, which means adding them to conventional production costs c'_q , as in Condition 4-2. The choice of abatement technology should reflect marginal damage costs $D'e'_a$, as in Condition 4-3.

$$P = c'_q + \sum_j D'_j z_{ij} e'_{iq} \quad (4-2^*)$$

$$c'_a = -\sum_j D'_j z_{ij} e'_{ia} \quad (4-3^*)$$

where z_{ij} is the transfer coefficient; D' , c' , and e' are derivatives of the damage, cost, and emissions factors, respectively; and subscripts q and a reflect output and abatement, respectively.

For stock pollutants, the rule implies equality of the marginal abatement cost not to the environmental damage today but to the discounted sum of the damages caused by current emissions in all future periods, taking into account their natural processes of decay or assimilation¹. If, on the other hand, the damage function of a flow pollutant varies with the temporal dimension (e.g., seasons or time of day, as mentioned in Chapter 22), then optimal marginal abatement costs also should be adjusted accordingly.

Direct Provision of Public Goods

The most straightforward “policy” an environmental protection agency can apply is to use its own personnel, know-how, and resources to solve a given problem. In the environmental arena, this mechanism is essentially the provision of public goods; whether the term policy instrument is appropriate is not clear (some economists would reserve “instrument” for policies that influence other agents), but it is important to start here.

Cleaning up public streets is a prime example of public goods provision; other examples include taking responsibility for major environmental threats, providing and maintaining natural parks, and managing certain kinds of research and control functions typically undertaken by environmental protection agencies. The U.S. Environmental Protection Agency, for instance, sponsors much research on environmentally friendly technologies. In some countries, ordinary sewage treatment or municipal waste management is provided as a public good, although the state or municipality usually tries to cover costs by charging some kind of user fees. Such fees may be part of property taxes or other taxes, or, as is becoming increasingly common, they may be user fees tied more specifically to the service provided (in the form of price-type instruments; see Chapter 5).

During the past couple of decades, in most countries, the state has started to refrain from acting as direct producer of goods and services. The role of the state can be broken down into several components: financing, administration, provision, and control. If the state finances an activity, it does not necessarily have to manage its provision. Several activities that were formerly thought of as natural state monopolies have been organized in such a way that the government agency merely retains a control function, and private entrepreneurs are hired to provide the services. Considering that the financing and provision of activities can be more or less decentralized means that public goods—including various environmental services—can be organized in many ways, of which direct public provision is one end of the spectrum.

The starting point for a discussion of the level of provision is the Samuelson rule, which focuses on the sum over all households of the marginal willingness to pay for a public good (see Chapter 2). The provision of roads, harbors, airports, railways, and telecommunications infrastructure has considerable impact on environmental quality. The decision criteria for these kinds of investments include issues of discounting, equity, and efficiency in the tax structure and public sector in general. In many countries, cost-benefit analyses are required for all major public investments, such as roads, and even on general reforms. The rules for such analyses may include a specified discount rate, time horizon, depreciation rates, accounting value of public funds, shadow prices for unemployed labor, foreign exchange, and similar crucial variables, including how to treat issues of income distribution. The way the rules of project evaluation are specified has a considerable effect as a policy instrument on the general path of development of the economy.

Regulation of Technology

One way of regulating the behavior of firms, households, agencies, and other agents in the economy is by prescribing the technology to be used or the conditions (through zoning or timing). The reasons why technology

standards, restrictions, and zoning are still the most commonly used instruments are their intuitive simplicity and perhaps the short time perspective of many policy decisions. These kinds of regulation may also suit the interests of both regulators and polluters.

Typically, complaints are made against a polluter that is using “unnecessarily” polluting technology. The proof usually is not only the pollution per se but also a comparison with other industries or to some proven (usually end-of-pipe, or chimney) technology. This approach has many names: *best practical technology*, *best available technology not entailing excessive cost*, *best available control technology*, or simply *best available technology*. This kind of instrument is often derisively referred to as “*command and control*,” which conjures up a rather Stalinist image, but these policies do allow for some trade-off. As the names suggest, technology must achieve a significant reduction in pollution, but at reasonable costs. The U.K. 1956 Clean Air Act used the term *practicable* and defined it (somewhat tautologically) as “reasonably practicable having regard, amongst other things, to local conditions and circumstances, to the financial implications and the current state of technology.” Regulations that restrict location or timing are *bans* and *zoning*: a ban is a form of technology regulation in which a specific process or product is not allowed; zoning is a kind of regulation whereby certain methods or technology (such as certain vehicles) are banned in or are limited to a certain area (see examples in Chapters 18, 20, 21, and 28).

The trade-off between costs and benefits is a sensible requirement. However, leaving this trade-off to be determined by a bureaucracy may not be so desirable. More flexible instruments allow enterprises to make the trade-off. With mandatory technology, enterprises have little choice and are not encouraged to explore cost-efficient ways of achieving pollution control. They cannot trade reductions between sources, and they are not given any incentive to develop cleaner technology. Instead, the “*best available technology*” concept tends to encourage end-of-pipe solutions. However, command-and-control mechanisms or standards are not necessarily always inappropriate. In some instances (such as mandatory catalytic converters), considerations other than efficiency (such as ease of inspection) may be an overriding concern.

To understand the operation of technology regulation as an instrument, consider a firm i maximizing its profit (π_i) as $Pq_i - c_i(q_i, a_i)$. The firm’s emissions [$e_i = e_i(q_i, a_i)$] are assumed to be perfectly mixed, and the regulator maximizes social benefit as in Box 4-1. With full information about abatement and damage costs, the regulator could specify the necessary individual technologies to achieve maximum welfare. In principle, both production technologies and abatement technologies could be used, but because end-of-pipe solutions are common in emissions regulations, the analysis is simplified by focusing only on abatement technologies and assuming that the regulator chooses the optimal individual level of abatement technology for each firm ($\hat{a}_i$) as its policy instrument² (as in Box 4-2).

BOX 4-2 Design Standards for Abatement

Firm i maximizes profit subject to its mandatory abatement technology ($\hat{a}_i$). The Lagrangean is

$$L = Pq_i - c_i(q_i, a_i) + \lambda_i(\hat{a}_i - a_i) \quad (4-4)$$

where P is product price, c is production costs, q is output, a is abatement, i is a firm, and λ is the Lagrangean multiplier. Taking partial derivatives gives the so-called Kuhn–Tucker conditions, which imply that if there is an interior solution (i.e., one in which the border conditions are not binding) with positive $(q_i^*, \hat{a}_i)$, then the following conditions hold:

$$P = c'_q \quad (4-5)$$

$$\lambda_i = -c'_a \quad (4-6)$$

$$a_i = \hat{a}_i \quad (4-7)$$

where the asterisk (*) denotes optimum and the circumflex (^) indicates a value given exogenously—in the case of $\hat{a}$, decided outside the firm, by the regulator.

The λ_i can be considered the shadow costs (in this context, roughly, the implicit costs) of abatement. With perfect information and if an environmental protection agency were able to select optimal (welfare-maximizing) choices of abatement for each firm ($\hat{a}_i$), then λ_i would be equal for all i (and equal to the effect of abatement on damages, $D'e'_a$, as in Condition 4-3).

Note: To include the possibility of a border solution where output or abatement is 0, equalities should be replaced with inequalities. The full set of conditions is

$$P \leq c'_q \text{ with equality if } q_i^* > 0$$

$$\hat{a}_i - a_i \geq 0, \lambda_i(\hat{a}_i - a_i) = 0, \lambda_i \geq 0$$

$$\lambda_i \leq c'_a \text{ with equality if } a_i^* > 0$$

Again, some subindices were dropped for convenience. For simplicity, only the equations for an interior solution with equality for all the other instruments are provided. They are simply referred to as the “optimality conditions.”

Under certain conditions, it is possible to achieve desired emissions levels through regulations on individual abatement (see Box 4-2). However, even with an optimal choice of abatement technologies, environmental damage is not reflected in the price, and thus output is not reduced as one of the mechanisms to reduce pollution. An individual firm cannot choose to meet the environmental target by searching for new (abatement or production) technologies or by reducing output.

Furthermore, it typically is not feasible for a regulator to have knowledge about individual abatement levels or technologies for each firm. The

information requirements and administrative costs are prohibitive. Typically, an environmental protection agency wants a standard end-of-pipe technology that is the same for all and easy to monitor (e.g., catalytic converters, filters, or chimneys). In this case, abatement and emissions levels typically will not be optimal. Firms have individual emissions functions, $e_i = e_i(q, a_i)$, which implies that the same level of abatement typically will not give the same emissions rates—and individual marginal costs of abatement (c'_a) typically would not be equalized. Therefore, the basic economic condition for allocative efficiency would not be fulfilled. If the marginal cost of abatement varies between firms, then money could be saved by moving some of the total abatement effort to the firms where abatement is (at the margin) cheaper.

The type of regulation analyzed above is not a regulation of emissions but of (abatement) technology. The design standard is a mandatory technology and leaves the firm almost no choice and is therefore bound to perform poorly with respect to flexibility. It typically also will fail to meet abatement and emissions targets because there is no regulation of output (see Chapter 11). Under certain conditions, the advantages of technology standards may still dominate. For instance, some of the following conditions might motivate the use of design standards:

- 1 Technical and ecological information is complex.
- 2 Crucial knowledge is available at the central level of authorities rather than at the firm.
- 3 Firms are unresponsive to price signals (e.g., because of a noncompetitive, transitional setting) and investments will have long-run irreversible effects.
- 4 The standardization of technology holds major advantages.
- 5 Of only a few competing technologies available, one is superior.
- 6 Monitoring costs are high: monitoring emissions is difficult, but monitoring technology is easy.

In real life, all these criteria will not be fully met, but in many situations, some of them are important. Presumably, this is why technology standards are still frequently used. One important example is mandatory catalytic converters on new cars, a regulation that is now almost universal (see Chapters 17 and 18). This example meets Criteria 5 and 6, and maybe Criteria 4 and 1. Catalytic converters have been “successful” in reducing emissions, and their almost universal adoption has lowered costs. Still, it is possible that some other combination of engine modification, alternative fuels, and traffic management would have been more cost-effective, at least in some settings.

Another example of specific mandated technology is the regulation of nuclear power plants. In case of an accident, damages would be high; thus, the goal is zero incidents. Plants are regulated not only by being given maximum emissions levels; they are also usually given considerably more detailed and

specific technology requirements, such as multiple separate control systems and certain kinds of containment. It can probably be best thought of in terms of Criteria 1, 2, 3, and 6 as well as the fact that society has absolved the firms from a large share of the risk. Cleanup after nuclear accidents usually is the responsibility not of the power companies but the federal government (see Chapter 10).

Technologies that are completely banned provide an example where the clarity of decisionmaking, economy of control, and ease of monitoring coincide. Examples are bans on certain kinds of chemicals (see Chapter 20), fuels or energy technologies, and vehicle types. Some of the issues concerning the potential advantages of being able to restrict bans to certain zones as well as the pitfalls of income distributional effects are illustrated in Box 19-1 in Chapter 19.

In natural resources management, the equivalent to technology regulation is mandatory technology (or restrictions on technology) for management, catch, hunting, farming, and so forth. It is (perhaps unfortunately) a fairly common kind of policy instrument. In fishing, the use of cyanide and dynamite is banned, which most people agree is reasonable (because these methods are so dramatically destructive to the actual habitat). However, techniques that are not necessarily destructive but perhaps simply more effective, such as those that involve enhanced nets and the use of equipment such as lights or sonar search equipment to attract or find fish, are also sometimes restricted (see Chapter 25). These restrictions are generally imposed to protect the resource but could in some cases be seen more as a way of protecting the livelihood and interests of those fishermen who use the older and more labor-intensive technology.

In agriculture and forestry, examples of technology regulation include the mandatory replanting of trees after harvest, mandatory construction of soil bunds and terraces to prevent soil erosion, and guidelines for pesticide and fertilizer use as well as other land-management practices (e.g., fallow periods or tillage technology) (see Chapter 26).

Regulation of Performance

Under many circumstances, mandated technology might appear favorable in a superficial sense but turn out to be unsatisfactory—or even fail—when viewed with hindsight. One unfortunate characteristic of much policymaking is short-sightedness. With the time perspective of a political cycle, a given technological fix might seem adequate. Time after time, experience has shown that technology can provide better solutions than those that are immediately apparent. An important condition is to provide some reasonable incentives to develop new technology. A mandatory technology—catalytic converters or filters to stop emissions or miles of hillside terraces to prevent soil erosion—is definitely not going to help in that respect. In general, it is more efficient to target emissions rather than technology, inputs, or output.

A regulation that imposes a certain limit to harvests or to emissions instead of requiring a particular technology is called a performance standard (as distinguished from a design standard). A performance standard regulates quantities and as such is sometimes referred to as a *command-and-control* mechanism. This categorization is somewhat unfortunate; performance standards are significantly different from mandatory technology because they give firms considerable flexibility in the choice of abatement method by which to meet the mandated goal.³ They also leave the firms a choice between output reduction and abatement level, and trade-offs between polluting units are possible. The formal modeling of this instrument is fairly similar to that of design standards, but the results are distinct. The regulator chooses maximum allowable emissions for each firm ($\hat{e}_i$), which optimizes within this constraint (see Box 4-3). Optimally chosen emissions limits imply not only that abatement costs will be allocated optimally but also that the product prices will reflect the costs of abatement.

Typically, the product price under a performance standard might be somewhat higher than with design standards and the quantity produced, in principle, somewhat lower. With performance standards, a firm has the additional flexibility to reduce emissions not only by abatement investments but also by reducing output. This flexibility usually is part of the socially optimal outcome because the cost at the margin of reducing output to lower emissions may be smaller than the costs associated with additional investments in

BOX 4-3 Emissions Limits or Performance Standards

The firm maximizes profit subject to the constraint $e_i = \hat{e}_i$, and the Lagrangean is

$$L = Pq_i - c_i(q_i, a_i) + \lambda_i[\hat{e}_i - e_i(q_i, a_i)] \quad (4-8)$$

where P is product price, q is output, c is production costs, a is abatement, e is emissions, i is a firm, and λ is the Lagrangean multiplier. The corresponding optimality conditions (see note to Box 4-2) are

$$P = c'_q + \lambda e'_q \quad (4-9)$$

$$c'_a = -\lambda e'_a \quad (4-10)$$

$$\hat{e}_i = e_i \quad (4-11)$$

The λ can be interpreted as shadow prices of pollution. With perfect information and an optimal (welfare-maximizing) allocation of permits ($\hat{e}_i$), the λ_i will all be equal to the marginal level of damage (D') in Conditions 4-2 and 4-3 (Box 4-1), ensuring optimal abatement and allocative efficiency. Compared with the corresponding Conditions 4-5 and 4-9, the product price now reflects the opportunity cost of emissions permits.

abatement. However, this flexibility does not exist if the emissions limits are expressed relative to output (i.e., pollution intensity), which is a situation that is somewhat intermediate between design standards and performance standards. With pollution intensity regulations, there is no output effect (similar to a design standard), but there is some flexibility in technology choice (similar to a performance standard).

In addition to the differences in monitoring costs and flexibility, other subtle differences exist between design standards and performance standards. In many real-world cases, industrial pollution is controlled by licensing procedures that are a mixture of set emissions levels (total or relative to output) and mandated technology. Licensing procedures do not allow for flexibility in attaining goals through trading between sources. Furthermore, the information and resources available to parties (industries and local or national authorities) who negotiate the individual standards may be asymmetric, often leading to fairly lax environmental standards. The fact that many firms in polluting industries appear to prefer licensing to market-based instruments reinforces this impression. If well managed by knowledgeable authorities, these negotiations may give reasonable outcomes (e.g., Brännlund et al. 1996 on “command and control with a gentle hand”). We will in coming chapters argue that cost heterogeneity is a fundamental reason for preferring the flexibility of market-based instruments. However often, cost heterogeneity in turn depends on differences in age and thus the vintage structure of an industry. Using this information wisely can allow the regulator to capture many of the flexibility benefits in well-crafted (phased) regulations; see for instance Carson et al. (2009) and Heutel (2010). However, one should be wary of the informational demands and the risks of corruption that this kind of regulation implies. The more judgment and flexibility exercised by the regulator, the better the potential result—but also the more expensive the regulation in terms of personnel. When regulators become familiar with the industries, conflict of interest is a risk. Incumbents always like licensing procedures because it gives many opportunities for rent-seeking. The opportunity to help formulate standards, requirements, and regulations also may give incumbents the opportunity to keep new entrants at bay.

One negative feature of controlling individual emissions rates (such as those set for new cars) is that they do not imply full control on total levels of pollution or ambient pollution levels. The total levels also depend on output for each agent and the number of agents. For example, with high mileages, “clean” (less-polluting) cars still lead to higher emissions even though the rate of emissions has been regulated, because the output (i.e., number of miles driven) has not. Even when all individual emissions (not the rates but the totals for each firm) are fully controlled, society does not have full control over aggregate totals. In the case of pollution, total emissions still depends on the number of polluters; thus, ambient pollution levels may be excessive.

In natural resources management, the total catch (or grazing pressure, or harvest levels) is the decisive factor for sustainability. Parameters are sometimes

set by formulating the quantitative limits, such as harvest quotas, as an individual share of harvest, rather than in terms of rates (see Chapter 25).

Monitoring is discussed from various viewpoints—enforcement, political economy, and moral hazard—in Part III. Monitoring a detailed vector of emissions is costly and often delegated to firms, but their data are not necessarily reliable. For ease of monitoring, regulators sometimes focus on the volume of inputs or some other factor that is readily controlled (e.g., number of fishing days or nets per person). One particular type of standard deserves special mention: level zero—that is, total bans. This particular level is particularly easy to monitor. With any level other than zero, inspectors have to pass judgment about the conditions pertaining to each individual case. Ease of monitoring may explain the popularity of bans, such as those on chlorofluorocarbons, dieldrin and other pesticides, cadmium, lead and other toxic paint pigments, arsenic, and the ivory trade.

Supplemental Reading

Arrow and Kurz 1970

Bohm and Russell 1985, 1997

Dasgupta, Marglin, and Sen 1972

Fisher et al. 2007

Keohane et al. 2009

Little and Mirrlees 1969, 1974

Meade 1951

Stavins 2006

Notes

- 1 Proofs appear in Kolstad 2010a, Baumol and Oates 1988, and Tietenberg 1992.
- 2 Abatement technology is assumed to be conveniently summarized by the variable a ; reality may be more complex. An environmental protection agency might require the use of some particular process, routine, or equipment that would need somewhat more complex modeling.
- 3 Flexibility is desirable because the firms are assumed to have the best overview of technical options and because these options vary between firms and over time. However, ease of monitoring is also an important goal. In the area of personal safety, for example, mandatory seatbelt use is not necessarily the best policy; it can encourage reckless driving and may have slowed the introduction of airbags and other constructional vehicle improvements. However, seatbelt use is relatively easy to monitor, and that factor might be enough to make it an optimal instrument overall.

Taxes

Economists often view environmental charges as the most expected instrument for environmental and natural resources policy and tend to use them as a point of reference for other instruments. A pure environmental charge is referred to as a Pigovian tax if it is set equal to marginal social damage (e.g., of some pollution) (see Box 5-1). At least under several classical assumptions (including fully informed, honest, welfare-maximizing regulators and appropriate concepts of property rights), they have certain optimality properties. However, in many cases, the pure environmental tax is hard to use (e.g., if pollution is unobservable) and the available proxies or substitutes (such as input or output taxes) are more or less suitable. In this chapter, we discuss some of the differences among these alternatives.

Pigovian Taxes

One of the main conditions for optimality is that a Pigovian tax is set equal to marginal damages, but these are difficult to estimate. The reasons for this difficulty include lack of understanding of the multiservice and public good characteristics of ecosystems (market prices commonly are missing, and a large subfield of environmental economics is dedicated to studying various methods of nonmarket valuation). Another complication is the fact that the damage curve may have a large slope. It is important that an environmental protection agency set the tax or charge equal to marginal damages at the optimal pollution level (i.e., at the intersection of the marginal damage and cost curves), which may be different from marginal damages at the time of the decision itself (e.g., see Figure 4-2 in Chapter 4). However, analogous problems exist for the setting of standards.

Particularly for economies with rapid growth (such as in many fast-growing developing countries and many of the formerly planned economies), the current or observed marginal damages may be difficult to calculate; evaluating their cost at a hypothetical optimum that society hopes to move toward is even more difficult. In a situation with high historic levels of pollution (as in

BOX 5-1 Pigovian Taxes

Consider a representative firm that seeks to maximize its profit after abatement costs and taxes. It solves

$$\max Pq_i - c_i(q_i, a_i) - Te_i(q_i, a_i) \quad (5-1)$$

where P is product price, q is output, i is a firm, c is production costs, a is abatement, e is emissions, and T is the Pigovian tax. The necessary and sufficient first-order conditions for a firm with positive output and abatement costs are

$$P = c'_q + Te'_q \quad (5-2)$$

$$c'_a = -Te'_a \quad (5-3)$$

where c'_q is the derivative of production costs with respect to output.

A comparison of Conditions 5-2 and 5-3 with the conditions for optimality in Conditions 4-2 and 4-3 shows that the conditions are identical as long as the Pigovian tax is correctly set equal to the level of marginal damages ($T = D'$).

many eastern European countries), the marginal damage may be expected to fall significantly as the worst excesses of pollution are cleaned up. Not overestimating the Pigovian charge may be crucial because in the long run, it might turn out to be too high, and in the meantime, high taxes may cause unnecessary firm closures, unemployment, and opposition to the environmental charge instrument per se (see Chapter 21).

Timing is crucial in the context of policy instrument implementation because environmental charges can cause liquidity problems for firms; taxes for firms are highest while their corresponding environmental investment costs are highest. One solution to this cash-flow problem is to levy a two-tiered tax on polluters: a zero- or low-level tax up to a certain “permissible” level, then a higher tax rate for “excess pollution.” Proponents of this scheme suggest that it makes polluters pay the appropriate tax at the margin while leaving firms sufficient funds to finance abatement investments (and avoid bankruptcy). A two-tiered tax would be appropriate for an intermediate concept of property rights to the environment. For example, a firm that has rights to some minimal level of ecosystem services would correspond to the part with zero or low payment. A firm that needs to pollute or use resources in excess of this level would have to pay the full rent for the resource use to which it does not have any rights. This principle is related to refunded emission payments and tax subsidies (see Chapters 7 and 12).

Other ways of avoiding liquidity problems include waiving taxes in return for substantial abatement investments, making the cost of certain investments deductible from the tax, or giving firms sufficient time to adapt by announcing the tax a few years in advance.

It is important to emphasize that a Pigovian tax (i.e., equal to marginal damages) is, under a range of circumstances and from a broad perspective, a full solution to the problem of how to internalize externalities. Such a tax is not only necessary but also sufficient, at least if appropriately adapted so that taxes vary with geographical and temporal variations in the damage function. The victims of pollution do not have to be compensated to arrive at an efficient resource allocation (Baumol and Oates 1988, Chapter 4; Oates 1983). Although (ethics- or rights-based) welfare reasons for compensating victims might exist, the practice may provide an undesirable incentive for the victims to locate next to polluters and intentionally not take steps to protect themselves against the effects of the pollution so they can collect compensation.

Taxing victims could, under some circumstances, be used to “keep the victims away” from such locations, thus reducing environmental damage, at least as conventionally measured by disutility to victims (Coase 1960). In general, this policy is not perceived as attractive or fair, but it does illustrate the nature of the problem. Consider, for instance, people who have intentionally located near major flood-prone rivers and then regularly collected compensation when the rivers flooded. The “tax equals damages” rule is enough to keep damages at the socially optimal level, and from the viewpoint of allocation at least, additional instruments (such as compensation) only distort the incentives away from the optimal level. However, in some cases, the victims are few and may not be able to do anything to increase or decrease the effect (e.g., through self-protection or through perverse effects of moving relative to the polluter’s location)—say, if the community decides to implement a technology that will disturb a small minority of particularly vulnerable people. In such cases, compensation probably will be motivated but should be designed in a lump-sum fashion to avoid perverse incentives for the victims.

Setting the level of tax is far from trivial; in fact, it is one of the main difficulties with the tax or charge approach. In addition to the purely technical difficulty of estimating slopes of damage and abatement cost curves (and their intersection), numerous informational and political issues must be addressed. The firms on which an environmental protection agency will rely for information usually have a tactical reason for either withholding or distorting information, and polluters (businesses, their trade unions, or even other public bodies such as municipalities, which are among the worst offenders) may exert considerable pressure on an environmental protection agency for lower (or no) taxes. In addition, because municipalities typically lack funds and play the dual role of polluter and regulator, they sometimes can grant themselves some form of immunity.

In many real-world situations, the charge is adjusted by trial and error. A form of *tâtonnement* (using trial and error to obtain an optimum solution; in this context, not only end-of-pipe investments but all investments where abatement may be only one small but perhaps inseparable component) can be emulated. However, this approach may be unsatisfactory, particularly when

abatement investments are expensive, and long-run stability is important for any policy that is to affect investments without undue cost. Introducing a given tax (T) in small, previously announced steps may help reduce resistance to the tax. Most important, this approach may leave the firms with sufficient funds to avoid the liquidity problems commonly associated with high environmental charges. However, “path dependency” may develop if low tax levels lead the firm to invest in minor end-of-pipe investments that later prove to be insufficient and thus unnecessary because the process has to be fundamentally changed as a result of higher environmental taxes. A fee or charge that is partly refunded (see Chapter 7)—or another instrument that implies smaller total (but the same marginal) costs to the polluter—might generate less resistance from polluters and could be set higher than would have been possible for a pure tax.

When damages are difficult to estimate, the abatement costs are sometimes used as a proxy. Although the two marginal costs are supposed to be the same in equilibrium, they both have slopes, and so a point estimate of one is unlikely to be the optimal value. However, if only the abatement costs can be estimated, this approach has the advantage of giving regulators a feel for the reasonable range of values and an idea of the financial consequences for the firms.

In some settings where regulation is the main instrument, fines are levied for noncompliance. Fines are commonly set equal to the “money gained” by not complying—which is roughly equivalent to abatement costs avoided. If the fine is the only punishment for noncompliance (and if detection is automatic), then this regulation can be seen and analyzed much like a charge, where the fine takes the place of the charge (see Chapter 13 for more complex cases with probabilities of monitoring and risk aversion).

Sometimes the difficulty in deciding the level of an environmental charge or charge tax is attributed only to the policy instrument, which is somewhat misleading. For instance, it may be asserted that an environmental protection agency must use trial and error to find the charge level appropriate to meet the target emission level. This assertion would be correct if the abatement target level was exogenously given and the only concern was to meet the target cost-effectively. However, the target cannot be determined without precisely the same information needed to find the charge level. One might just as well speak of an optimal charge level (which might be based on a fairly reliable estimate of how much the companies can “afford”—or will accept without migrating or fighting the tax in the courts) and trial and error with quantitative limits to achieve the corresponding abatement cost.

From the viewpoint of efficiency, the determination of the two variables (fee and quantity) is simultaneous and symmetric. However, in practical environmental protection, target levels may be formulated by ecologists or engineers (rather than economists) and gain general acceptance without being explicitly deduced from an analysis of marginal costs of abatement and environmental damage. In situations with strong irreversibilities or threshold values, this approach may in fact be reasonable (instruments used under conditions of

uncertainty are compared in Part Three). It also may apply to national policies formulated to meet obligations under an international agreement.

Taxes, Charges, and Earmarking

The terms taxes and charges are sometimes used interchangeably in this and other texts. However, taxes usually are reserved for politically rather than administratively decided fees and typically go to the treasury rather than being earmarked for local or sectoral use, whereas charges may be levied and appropriated by sectoral agencies. In principle, this distinction is important, but some hybrid or borderline types of fees also exist, such as politically decided fees that are earmarked for environmental funds and locally decided fees that are paid to the general treasury. The previous analysis applies to the classical Pigovian tax. However, taxes have a couple of disadvantages, one of which is the relatively complex legal process involved in passing and modifying tax laws, which can make the tax instrument somewhat blunt.

Furthermore, many politicians have encountered considerable resistance to environmental taxes, and local or sectoral charges (over which the polluter or resource user has some influence) are typically more readily accepted (see Chapters 11 and 12). In many countries, money that goes to the central treasury is perceived as being “lost,” whereas money that stays within the sector or region is not. Environmentalists, too, commonly favor charges that are earmarked for publicly financed abatement. Some even view the main purpose of the fee as to raise money for such purposes, which misses the central points of a Pigovian tax (i.e., the incentive for abatement and the output substitution effect).

In general, economists are skeptical of earmarking taxes for special purposes, typically arguing that all tax revenues should go to the treasury and that public goods (including abatement) should compete on an equal footing for public funds. In both developed and developing countries, earmarking is considered an additional constraint in the optimization of government taxes and expenditures (OECD 1996; McLeary 1991). With such an approach, it never can be the optimal policy. Other approaches—such as public choice or asymmetric information—are required to allow for the possibility of earmarking as a (“second-best”) strategy.

In general, the financial tradition in Europe is negative to earmarking or “hypothecation.” However, in the United States as well as many other industrialized and developing countries, funds are commonly earmarked for such major economic undertakings as road construction. In an economy where many funds are already earmarked and where the public allocation of funds does not operate optimally, political economy arguments may favor the earmarking of some environmental fees, too. An analogous situation in environmental policy relates to water and sewage management, for which fees are charged to finance supply. Similarly, the money from fishing and hunting

licenses commonly is used to preserve habitat. The U.S. system appears to be more in favor of earmarking not only fees for road construction but also environmental charges, as illustrated by Superfund, the Oil Spill Liability Trust Fund, and other funds to which most U.S. environmental taxes are paid. Despite the generally negative attitude toward earmarking (e.g., of fuel taxes) in France, water charges and emission taxes on sulfur, volatile organic compounds, and nitrogen oxides are earmarked for abatement investments by the polluting firms (e.g., OECD 1995).

If one accepts some form of earmarking, then one reasonable case might be to use sulfur taxes to finance the liming of lakes because it helps redress their ability to withstand the acid rain caused by sulfur emissions. Although it would be pure coincidence if the optimal allocation for liming happened to equal the proceeds from a sulfur tax, it may still be a politically convenient arrangement. An early proponent of earmarking suggested strong public choice arguments in favor of earmarking as a flexible mechanism for allocating public funds (Buchanan 1965). Distributional concerns may also be important. If all the taxes are raised in one urban area, for example, then it seems realistic to expect local pressure to keep the proceeds rather than spread them across the whole nation (see Chapters 21 and 28). Similarly, if a certain group of people (such as motorists) pays more taxes, then that group will pressure for spending that benefits the group. In the real world, the government poorly distinguishes between agents of the economy. Particularly when environmental taxes have large potential proceeds (such as those resulting from a carbon tax), the distributional effects of taxation and spending may be significant. In such cases, the general equilibrium effects of the proceeds must also be considered (Chapter 11). In the absence of optimal tax and transfer instruments, earmarking may be a second-best strategy (Pirttilä 1998).

Taxes on Inputs and Outputs

When emissions monitoring is impossible, difficult, or costly, taxes may be levied on some input or output that is more easily monitored and a good indicator of (or proxy for) the pollution to be regulated. For the sake of simplicity, taxes on inputs or outputs are sometimes included among the environmental taxes, but they are actually somewhat different. They sometimes are referred to as “presumptive taxes” because, in the absence of direct monitoring, the agent that uses a certain input or produces a certain output is presumed to be polluting. Because monitoring individual emissions is often impossible in developing countries, some researchers suggest that presumptive taxes should be used—for instance, taxes based on inputs or outputs that can be presumed to be close complements to the pollution. A polluter that demonstrates abatement or clean technology can be exempted or refunded, but the burden of proof is moved from the environmental protection agency to the firm

(Eskeland and Devarajan 1996). The importance of this factor is presumably greatest in poor developing countries that have little experience with environmental control. Excise taxes on products such as cars, fuel, and cigarettes already make up a large share of taxation in these countries (Sterner 1996). Although environmental motives may have played some role in creating these taxes, the main reason is probably the ease of tax administration; another factor may be the relative attractiveness in taxing these goods as luxury goods, or for moral or ethical reasons.

If the output of a firm is closely correlated to the pollution to be regulated, then the product tax may be a good proxy. For example, consider a firm that produces hydrochlorofluorocarbons (HCFCs; substitutes for chlorofluorocarbons that are somewhat less damaging to the ozone layer). An output tax would be about the same as a tax on ozone depletion, although there should be a deduction for HCFC recycled or collected in closed systems. The workings of an output tax are illustrated in Box 5-2. A similar approach applies to the use of inputs that are directly related to the amount of pollution. Thus, if a firm uses HCFCs as an input, then an input tax is almost the same as a tax on ozone depletion (unless the firm collects and disposes of the chemicals appropriately, in which case the tax ideally would be rebated).

If absolutely no abatement technology is available (which is highly unlikely), then a product tax will be the optimal response because it incorporates the externality into the product price and thus allows the demand side to

BOX 5-2 Environmentally Motivated Product or Output Taxes

Consider a firm as in Box 5-1, with the difference that there are no abatement measures available. This can be modeled by replacing $e_i(q_i, a_i)$ from Equation 5-1 with $e_i = \xi q_i$, where e is emissions and ξ is a constant. This substitution implies that there is no abatement (a), giving

$$\max Pq_i - c_i(q_i) - T\xi q_i \quad (5-4)$$

where P is product price, q is output, i is a firm, c is production costs, and T is the Pigovian tax. There is now only one first-order condition:

$$P = c'_q + T\xi \quad (5-5)$$

where c'_q is the derivative of production costs with respect to output.

The welfare function (Equation 4-1) that the social planner seeks to maximize becomes $W = \sum_i [Pq_i - c_i(q_i)] - D(\sum_i \xi q_i)$, giving the condition $P = c'_q + \xi D'$, where D' is marginal damage (compare this condition with Condition 4-2; Condition 4-3 does not have a counterpart because there is no abatement). As expected, this result is achieved by setting the tax equal to marginal damage $T = D'$. Because the emission coefficient is constant, the emissions tax can be replaced by an output tax of $T\xi$ per unit of output.

give an “output” effect (i.e., reduced output as a result of consumers choosing other goods). However, if the possibility of some type of abatement exists, then an output tax will be suboptimal. If the abatement possibilities are severely limited and monitoring is difficult, then taxes on inputs or outputs may be good second-best instruments. Gasoline taxes are one example. Because of its flammable nature, gasoline is sold subject to a certain amount of physical control in most countries, so the extra (administrative and technical) cost of taxing gasoline sales is small. The relationship between gasoline consumption and the environmental damage caused by vehicle emissions is far from simple, but valuing and monitoring emissions is so complex that gasoline taxes are in fact often chosen as a proxy. In this case, input taxes do not provide any incentives for technical progress or substitution that could help in abatement, and therefore, it may be advantageous to combine gasoline taxes with other instruments, such as mandatory catalytic converters or even subsidies for such abatement technology (see Chapter 21).

Presumptive taxes provide an incentive for clean firms to disclose their technology if they can thereby reduce their fees. One good example is the Swedish tax that is levied on the sulfur content of fuels. This sulfur is presumed to be released on combustion. However, the tax is repaid on proof of abatement (see Chapter 20). This method is somewhat akin to a deposit–refund system (see Chapter 7) or a so-called tax expenditure—an exception to taxation that may be seen as an implicit subsidy for some kind of activity. Tax expenditures are quite common in OECD as well as in developing countries (Stern 1993, 1996; see also Chapter 21).

Taxing Natural Resources

In natural resources management, the counterpart to Pigovian taxes is the levying of fees such as mining royalties, stumpage fees, user fees, and land taxes. These fees might be charged because of external effects (including stock or catchability externalities) or because of the absence of property rights and thus of payment for a scarcity rent (1) as in the Hotelling rule (see Chapter 22). Policymaking consists of correcting for externalities and other market failures as well as imposing a scarcity rent when there is no effective owner who can claim such a rent. The analysis is closely analogous to the case of depletable minerals discussed in Chapter 22; however, for many renewable resources (such as forests, water, fish, and other ecosystem resources), the dynamics are more complicated. Fees charged for natural resources are commonly a considerable source of revenue in low-income countries, and in other countries, they could be (some examples are discussed in Part VI).

Supplemental Reading

Baumol 1972

Bovenberg and de Mooij 1994

Cropper and Oates 1992

Diderik 2009

Gaudet 2007

Metcalf 2003

Muller and Mendelssohn 2009

Tradable Permits

One logical way to control aggregate levels of emissions or harvest is to set a total number of permits or quotas to adapt to the assimilative capacity of the environment or the sustainable harvest yield, respectively. This can be seen as the direct creation of property rights or “enclosure” discussed in Chapter 3. Setting totals while allowing for some dynamics in the economy due to population growth, changing technology, mobility, and economic growth means that the allocated permits must be transferable. Otherwise, the allocation of all available rights would make all new activities impossible by definition. Tradability also allows the efficiency of the market mechanism to be harnessed to ensure that marginal benefits and costs are equalized.

The resulting instrument is called tradable emissions permits (TEPs) in pollution management and individual transferable quotas (ITQs) in fisheries management. Similarly, transferable grazing rights, development rights, and other mechanisms apply to other areas of natural resources management. The theoretical foundation of this instrument is Coasian (Coase 1960); however, such mechanisms are generally ascribed to Dales (1968a, 1968b), although others had similar ideas (e.g., Crocker and Woloizin 1966). Dales’ suggestion was to create an authority in Ontario, Canada, that would sell “rights to pollute” water bodies. A local authority would thus decide total pollution, and the market would allocate the pollution rights among firms to reflect their demand for pollution or their abatement costs. The efficiency of this mechanism has been demonstrated (Montgomery 1972), and the instrument appears to have evolved just as much from the actual experience of regulation as from academic analysis. As discussed in Part I, trading follows naturally after property rights have reached a certain level of maturity.

The creation of tradable permits helps remove the externalities implied by the absence of property rights or the “public good” character of the environment. Essentially, this mechanism creates property rights to new resources or shares in the assimilative capacity or the sustainable rent production of ecosystems. We are witnessing a historic process of “enclosure.” More and more aspects of the ecosystem are becoming scarce, and, in response, society develops new kinds of property rights. The fact that these property rights

BOX 6-1 Permits as an Economic Instrument

With information on aggregate abatement and damage curves, regulators determine the socially optimal level of aggregate emissions ($E^* = \Sigma e^*$) and issue a corresponding number of permits. These permits are allocated so that each firm receives e_{i0} permits. Each firm is free to choose whatever combination of production, abatement, and permits it wants—subject to trading constraints, because it must hold a number of permits equal to its emissions level.

The firm maximizes profit subject to this constraint, as in

$$\max Pq_i - c_i(q_i, a_i) + p_e[e_{i0} - e_i(q_i, a_i)] \quad (6-1)$$

where P is product price, q is output, i is a firm, c is marginal cost, a is abatement, p is the price of permits, and e is emissions. The necessary and sufficient first-order conditions for a firm with positive output and abatement costs are

$$P = c'_q + p_e e'_q \quad (6-2)$$

$$c'_a = -p_e e'_a \quad (6-3)$$

A comparison with the corresponding Equations 4-5, 4-6, 4-9, and 4-10 and Conditions 4-2 and 4-3 for optimality (all in Chapter 4) shows the similarity between the sets of conditions. Conditions 6-2 and 6-3 should imply that p_e is equal to the marginal damage of pollution (D') (and thus the optimal Pigovian tax rate), which also implies that Condition 6-2 is identical to Condition 4-2 and that the output price under tradable permits correctly internalizes environmental damage.

Note: Optimality requires the correct number of permits and conventional assumptions concerning the behavior of firms (e.g., Baumol and Oates 1988). Both tradable emissions permits and taxes provide for a socially optimal long-run allocation when entry and exit are free (e.g., Spulber 1985; Xepapadeas 1997). Notice that for a new entrant, Equation 6-1 is modified, because $e_{i0} = 0$.

internalize externalities and create incentives for protection means that resources have a good chance of being put to their most efficient use (see Box 6-1).

Severe conceptual and practical problems must be overcome for the permits or quotas to work. They must acquire the characteristics of property rights, such as permanence and reliability. It takes time and commitment to develop permanence and trust, and in the case of natural resources management, a lack of knowledge about the underlying ecosystems and lack of agreement about how they should be managed create additional difficulties. Many features of permits cannot be decided with only ecological and technical calculations. (The problem is analogous to setting the optimal tax level.) They include the definition, number, duration, and temporal and spatial validity of the permits as well as the proposed method for their allocation. These decisions are crucial, partly because permits involve the transfer of essential property rights and, potentially, some substantial transfers of wealth. With policy

instruments such as licensing and technology standards, gradual calibration of the instrument to gain information and adaptation to changing circumstances can be imagined. Such changes will always meet some resistance from polluters, but they are possible. The creation of permits or quotas with the characteristics of property rights will exacerbate the difficulty due to their irreversibility.

Thus far, permits have taken two main forms. The early U.S. programs traded in credits that were created by overcompliance with other preexisting legislation (standards or emissions rates); later cap-and-trade programs (discussed later in this chapter) traded in rights that were created expressly for this purpose. Cap-and-trade programs have reduced administrative burdens and thus have encouraged the formation of real permit markets. Because the mechanism is so simple and builds on actual measurements and on predetermined permit allocations, the need for and the possibility of any discussion concerning baselines, exemptions, and so forth between regulators and firms is removed. Transparency and accountability reduce transaction costs and inspire confidence in the mechanism. However, modifying the overall level of permits would be difficult unless the permits are formulated as shares in a total that can be determined from year to year. (This procedure is common in fisheries, which are characterized by extreme variability in total allowable catch.)

Property rights must convey a sense of permanence to have an effect on costly investment and business decisions. However, this permanence is a problem for an agency struggling with severe information problems and desiring to keep some flexibility, for example, to adapt to changing conditions or new information. Optimal values for total emissions or harvest may vary considerably between locations as a result of ecological, technical, and socioeconomic conditions, and over time—both cyclically (depending on time of day, season, and weather) and as a result of changing patterns of economic activity, technology, and demographic patterns. They also may vary in response to underlying changes in dynamic complex ecosystems. Many industrial and agricultural processes do not emit a permanent or predictable flow of a particular pollutant; however, they can be described as posing a certain risk. In the best of cases, the probability distribution of this risk might be known, and the parameters of that probability distribution might be the target of change.

How can these uncertainties be handled in a way that permits some flexibility to the decisionmaker while providing sufficient certainty to the permits? One way is to limit tradability over time. This approach has been used for pollutants that were expected to be reduced rapidly over time (as a result of technical progress in abatement) and in cases where there was considerable uncertainty about the damage function. In such cases, the permits may have a limited life, so permits for every period have to be acquired during that period with no (or limited) banking possibilities. The allocation of permits might still be proportional to allocation in previous years (a form of grandfathering).

Another mechanism that combines flexibility with certainty is conceptually similar: allow the total number of permits to vary depending on ecological

or other conditions. Fisheries are a good example of this mechanism because the dynamics of the underlying ecosystem are particularly difficult to predict beforehand. As a result, tradable harvest permits for fish are often formulated as quotas—that is, shares in total allowable catch (see Chapter 25).

This component of design gives decisionmakers flexibility while still giving a reasonable degree of property rights to the firms. The need for this flexibility is readily apparent in fisheries, but society might be wise to ensure a similar type of flexibility in other areas. Naturally, industry needs a high degree of certainty to undertake cost-efficient investments, which requires knowing that permits correspond to certain physical quantities for more than one year. This need could be addressed by an environmental protection agency committing itself to revisions of total emissions that are infrequent and occur only on fixed, announced dates.

U.S. Emissions Trading Programs

This section is dedicated to emissions trading programs that have been operated in the United States: “bubble” policies; ambient permits; the output-based allocation of permits used for the phaseout of lead from gasoline (which is of particular interest because it is a different mechanism); and the cap-and-trade programs that are currently used, for instance, to abate sulfur emissions, nitrogen oxides, and GHG emissions.

Emissions Reduction Credit Programs

The history of TEPs is instructive. Their beginning appears to have been as an extension of the system of emissions limits for air pollution within the Clean Air Act. In short, the polluters argued for flexibility and persuaded the regulatory authorities that they could achieve the same or even more abatement if they could allocate abatement efforts themselves and if they could trade reduction at one source that was more than required for incomplete compliance at another. This approach opened up the possibility of economic growth in areas of nonattainment and was the start of what came to be known as the Emissions Trading Program.

The Emissions Trading Program, initiated by the U.S. Environmental Protection Agency in the late 1970s, built on a preexisting detailed system of source-by-source regulations and individual pollution permits or licenses (discussed in Chapter 4). The problem was that in certain areas of the country, these policies were not sufficient to attain the goals set for ambient concentrations of various pollutants. Such policies cannot be relied on to meet aggregate or ambient targets. These so-called nonattainment areas continued to have unacceptable levels of pollution, and progress toward meeting ambient goals was considered much too slow. Generally speaking, these areas had high

population density and large concentrations of industry. Economic growth was a top priority, and the last policy they wanted was one by which industrial expansion—hence job opportunities—would be lost.

Yet such was public perception of the policy. In some cases, policy even appeared to give the worst possible outcome by being a hindrance to industrial output without being able to provide a clean environment. Policy reform was urgently needed, and the basic ideas behind the Emissions Trading Program were thus to provide flexibility to firms so that they could meet environmental restrictions in the most cost-effective way, which might mean reducing emissions at a source other than the one targeted in the individual licenses, and allow industries to build new plants (or expand existing facilities) without increasing total emissions.

Several somewhat separate mechanisms were used—netting, banking, offsets, and bubbles—but all stemmed from the emissions reduction credit (ERC), which was an “excess” reduction that sources would voluntarily undertake, thereby exceeding their legal abatement requirements. The sources could apply to the control authority for certification of the ERCs, which then became the legal tender of the program, so to speak. The credits could not be traded freely, but trades were allowed under several programs:

- The offset policy allowed new firms to start up (or old firms to expand) facilities, even in nonattainment areas where otherwise no new plants would be allowed. To do this, the firm was required to not only comply with stringent regulations but also to buy ERCs from existing firms in the area, so that the regional aggregate emissions would be no greater after the expansion or establishment than they were before.
- The bubble policy applied to existing emissions sources in an area and allowed them to act as one unit for the purposes of compliance with emissions regulations. Thus, a firm could reduce emissions at some location less than would normally be required by regulations if it simultaneously reduced emissions more than would normally be required at another location. Because compensating ERCs could be from other firms, this program provided room for emissions trading.
- Netting allowed existing firms that wanted to expand or modify the option of avoiding the stringent review process for new sources as long as the net increase in emissions (after reduction of ERCs acquired) was below a certain threshold.
- Through banking, firms were allowed to store certified ERCs for subsequent use in any of the above programs—i.e., for bubbles, netting, or offsets.

All of these mechanisms required buying “extra reductions” from the regulated firms in the region that were already fulfilling their legal obligations but could perhaps be motivated to achieve more abatement. The persuasive factor was the returns from selling their excess permits, which underscores

the importance of the permanence of these permits as rights. A naïve critique of the program would claim that environmental improvement at one plant should be seen as only an environmental improvement and that the firm in question should not be allowed to sell extra credits to another firm, because then pollution is constant. This critique completely misses the point, because in this setting, firms are already regulated; additional abatement is costly and hence will be undertaken only if there is some possibility of profit, or at least cost recovery. The abatement effort that results from permit trading leads to a cleaner environment—at least relative to the ambient level obtained with the same industrial output and equivalent mandatory standards—which is the goal of the regulation. Alternatively, the policy instrument could “accommodate” more industrial output for any given level of ambient standards and total abatement costs.

In a situation where pollution is essentially unregulated, handing out permanent emissions rights would not be without risk. If many sources that can easily abate obtain large amounts of emissions rights, then incentives for abatement at other sources are reduced. The latter sources can simply buy excess permits instead of abating emissions. (This problem has been referred to as “hot air” in the context of global warming.) In extreme cases, it is conceivable that the disadvantages of this allocation of excess permits would be a stronger negative influence on pollution than the benefits associated with the economic incentives to abate created by the system. Particularly in developing countries, where some polluters may have essentially unregulated emissions, care should be taken in setting up trading systems to avoid giving away excessive rights permanently.

In many of the first U.S. programs, considerable cost savings were achieved, but the theoretical potential (worked out by comparing current regulation with an idealized model that results in equalization of marginal abatement costs, for instance) was seldom realized. Several programs had very little trading, and transaction costs proved to be important. Individual approval by the appropriate authorities of each trade was often a considerable practical problem. There also was insufficient confidence in the ERCs as secure property rights (see Chapter 20 for a discussion on cost savings in sulfur abatement). Issues of design that could be sensitive to encouraging entry or delaying exit also turned out to be crucial (see, e.g., Stavins and Whitehead 1997).

Ambient Permit Trading

One problem of permit trading is that it may create regional or local “hot spots” (i.e., subregions with particularly high concentrations of industry and pollution). Permit trading can accommodate economic growth while maintaining total emissions within a country or specified region at a given level. This level might be constant or could decrease over time in response to expected technical progress or shifts in demand. The area covered must be

large enough to allow trading between polluters with varying costs of abatement if there are to be any cost savings. On the other hand, it must not be too large (and heterogeneous), because then damage levels might vary considerably, and in some spots, pollution might be particularly bad (“hot spots”). In models with multiple receptors and multiple sources (discussed briefly in Chapter 4), the transfer coefficients from source i to receptor j (z_{ij}) must be taken into account. Otherwise, there is no guarantee that the trading program will not worsen ambient pollution levels at hot spots.

The problem of hot spots can be solved by having separate permit trading systems for each relevant area or by a spatially differentiated ambient permit system (APS). The APS is optimal if its transaction and operational costs are disregarded. APSs are complicated because polluters have to acquire different permits (in proportion to emissions and transfer coefficients e_i and z_{ij} , respectively) for every receptor area. One simplification is to have an emissions permit system defined for the whole area but to make all individual trades subject to approval so as to guarantee that ambient levels do not deteriorate in prioritized hot spots. Alternatively, the emissions credits could be given different values in different areas, creating differentiated “exchange rates” for trades between areas or zones (depending on how sensitive they are). This approach is referred to as zonal trading and is the simplest of the APS schemes. The trouble is that these modifications remove much of the attraction of the system. The permits in the areas with lower concentrations of industry and pollution would fall in value if they could not be exchanged freely for other permits. As the complexity of the system increases, transaction costs increase, and the whole perception of permits as attractive property rights erodes.

The difficulties of APSs are particularly aggravated if the rules are modified *ex post* rather than being made explicit at the launch of the program, which severely restricts the possibilities for environmental protection agencies to adapt their regulations to new conditions. In the case of an inexperienced protection agency or a new pollutant for which uncertainty is great, tradable permits might be considerably less efficient than nontradable regulations.

The practical difficulties are aptly illustrated by the fact that full-fledged APSs have not been implemented anywhere. In fact, their counterpart—regionally varied taxes or fees—also involves many difficulties and is rare (however, see the Chinese example in Chapter 21). On the whole, it seems as if the difference in damage has to be considerable before the extra cost of spatial differentiation of the policy instruments will be justified. One of the best examples is the use of offsets in the United States, where new sources were allowed in nonattainment areas only if they could buy ERCs from existing sources in such a way that the total ambient levels in the relevant area were improved (or held constant) (Foster and Hahn 1995; see also Chapter 20 on sulfur trading).

Output-Based Allocation and the Phaseout of Leaded Gasoline

EPA established limits to the lead content of gasoline since the early 1970s. Lead limits were enforced on a refinery-by-refinery basis; each refinery was allowed to average lead concentrations across the gallons it produced. Nevertheless, in 1982 the rules were changed to allow trading across refiners. Lead rights were allocated to all vendors in proportion to their gasoline sales at the refinery and retail levels.

Under the trading scheme, a refinery could produce above the limit if it purchased a sufficient number of rights from refineries that produced lead below the lead limit; that is, “inter-refinery averaging.”

This mechanism implied an output-based allocation of rights—that is, a regulation of pollution intensity rather than of total amount of pollution. This feature is analogous to a standard for pollution intensity, with the important difference that the rights were tradable. It is also somewhat similar to the operation of refunded emissions payments (see Chapter 7). Output-based allocation sometimes is referred to as benchmarking because the firm must compete against a benchmarked rate of pollution intensity, so to speak. One effect of tying permit allocation to output rather than auctioning or grandfathering permits is the lack of a role for output substitution. Another important difference is that the cost distribution among polluters is different and more favorable to new, clean firms than to firms that historically have emitted the largest share of pollution (and thus are favored by grandfathering). This difference can be decisive in securing political acceptability of the instrument.

With a fixed allocation of permits, an incentive to limit the production and consumption of the final good into which the pollution is embodied is intrinsic to the system, because the marginal cost of abatement (which, in equilibrium, is equal to the permit price) is part of the product price. However, with an output-based allocation system, no such incentive to reduce output exists, because increased output leads to more pollution rights. The system is theoretically less efficient than a cap-and-trade permit system (or a Pigovian charge on lead) in this respect. Nonetheless, it may be preferable in cases when the output effect is not desirable but decisionmakers want to affect abatement technology (see Chapter 7 for more analysis).

Cap-and-Trade Programs

During the 1990s, U.S. policymaking moved gradually toward cap-and-trade programs, which build on the earlier programs but emphasize tradability, security of property rights, and total emissions levels. The name of the programs is indicative of the procedure followed. First, regulators determine the total quantity of allowable emissions or resource harvest (the “cap”). Next, rights to those emissions are allocated among polluters (by auction, grandfathering, or some other mechanism but not relative to current output, because the total

amount of emissions is capped). Then, the allocated rights become subject to free trading among the polluters that own the rights. Cap-and-trade programs are the most flexible programs for emissions trading to date, and they best correspond to the idealized TEP described in Equation 6-1 and in Condition 6-2 and Condition 6-3 because they also regulate total emissions.

In a somewhat Coasian way, the initial distribution of permits in a cap-and-trade program (whether by auction, grandfathering or some other mechanism) is thought to have little effect on allocative outcomes. Shares, once distributed, can be seen as a windfall to the companies that receive them—or to their shareholders.¹ Because the permits are tradable, the owner has to decide whether to sell (or buy) or retain permits; at the margin, an extra unit of pollution implies a cost to the firm. In this sense, the marginal damage of pollution (the opportunity cost of the permits) becomes part of the marginal cost of production to the firm. Hence, once allocated, the permits act much like property rights to the environment. In fact, the effect of permits on product prices is much like that of a Pigovian charge, because the final product price reflects the cost of environmental “rent.” This observation assumes that firms treat permits as liquid assets with an opportunity cost and that they do not hoard them strategically to keep competitors or neighbors away, as they might do in monopolistic competition. Furthermore, there is the important issue of whether the allocation of permits is permanent, or if not, how often and in what way the system will be updated. If the system is updated based on performance indicators (such as emissions or output), then incentives for performance may be affected more or less strongly depending on the exact construction chosen. Similarly, issues such as how to allow newcomers into the system may have incentive effects on output levels or on the entry and exit of firms.

The most important cap-and-trade programs to date are the sulfur oxide (SO_x) and nitrogen oxide (NO_x) emissions trading programs in the United States. The sulfur trading program has generally been considered a great success. Several studies indicate that benefits have exceeded costs by a very significant margin, permit trading has been lively, and the price of permits has fallen below initial forecasts, indicating that comparative advantages in abatement have been significant and exploited and thereby lowered the costs of compliance (see Burtraw and Palmer 2003, Chestnut and Mills 2005, Napolitano et al. 2007, Burtraw and Szambelan 2009, and Chapter 20).

RECLAIM is a program that targets NO_x or SO_x emissions (the main source of smog in the Los Angeles area) from permitted equipment. Participating firms receive trading credits based on past peak production and according to existing rules and control measures. Credits are assigned each year and can be traded for use within that year. RECLAIM has become known for some considerable price volatility in permit prices. For instance, permit prices soared from some US\$500 to US\$60,000 in the third quarter of 2000; consequently sources began to fall into noncompliance, calling program design into question.

There is strong evidence of the close connection between regulatory uncertainty and structural shifts in allowance market prices. In recent years, NO_x prices have fluctuated in response to legal and regulatory uncertainty (Burtraw and Szambelan 2009). The ultimate source of uncertainty in the long run is the development of climate policy. Since regulatory action to reduce CO_2 will have a direct effect on the cost of operating power plants, it will also affect the performance of the markets for SO_2 and NO_x in the United States.

U.S. Carbon Markets

Although the United States pioneered emissions trading, and although it is by far the industrialized country with the highest per capita emissions, the development of a national carbon market has been slow to emerge. In October 1992, President George H. W. Bush supported action on GHGs, signing the UNFCCC treaty. However, in July 1997—before the Kyoto Protocol was finalized—the Senate unanimously passed by a 95-0 vote the Byrd-Hagel Resolution, which stated that the U.S. Senate would not ratify any protocol that did not include binding targets and timetables for developing as well as industrialized nations or that would result in serious harm to the U.S. economy (Bang et al. 2007). The Byrd-Hagel Resolution limited the U.S. negotiating position in the Kyoto Protocol. U.S. negotiators were not able to secure quantitative commitments for developing countries, which prevented the Clinton-Gore administration from putting the Kyoto Protocol to a vote in the Senate. In spite of this failure, Vice President Al Gore symbolically signed the Protocol in 1998, indicating that the Protocol would not be submitted to the Senate for ratification until there was participation by the developing nations. Finally, in 2001 President George W. Bush's repudiation of the Kyoto Protocol made clear that the U.S. Kyoto targets would not be implemented during his administration. (See Stavins 2008 and Brohé et al. 2009 for a detailed description of U.S. climate policy.)

Though the Bush administration consistently opposed any kind of mandated limitation on U.S. GHG emissions, favoring instead initiatives that relied only on voluntary cooperation and technology subsidies, a number of legislative cap-and-trade proposals came forward at the regional level. A wide variety of cap-and-trade proposals have been discussed in Congress, and other initiatives are under way (see Chapter 20). For instance, in 2009 there were three regional cap-and-trade programs in operation or development within the U.S., accounting for a total of 26 states and 36% of total emissions. Also, the Chicago Climate Exchange (CCX), a private and voluntary market for emission allowances between firms, has been founded.

The Regional Greenhouse Gas Initiative (RGGI) was the first mandatory U.S. cap-and-trade program for CO_2 . In 2007, the northeastern and mid-Atlantic states that are participating (Connecticut, Delaware, Maine, Maryland, Massachusetts, New Hampshire, New Jersey, New York, Rhode Island, and Vermont) set a cap on CO_2 emissions from fossil fuel-fired power

plants of at least 25MW, covering in total about 225 facilities. Their goal is to reduce CO₂ emissions from the power sector 10% by 2018. Participating states negotiated statewide caps largely based on historical emissions. Aggregated, these state caps form the regional RGGI cap. Interestingly, most states are auctioning 100% of emission allowances to industry and using the revenues for programs that promote renewable energy and energy efficiency or mitigating possible increases in energy prices to consumers. Covered entities are required to continuously monitor and report their emissions. Penalties for noncompliance will be enforced according to each state's rule.

The Western Climate Initiative (WCI) began in February 2007 when the governors of Arizona, California, New Mexico, Oregon, and Washington signed an agreement directing their respective states to develop a regional target for reducing GHG emissions, participate in a multistate registry to track and manage GHG emissions in the region and develop a market-based program to reach the target. The WCI built on existing GHG reduction efforts in the individual states as well as two existing regional efforts. In 2003, California, Oregon, and Washington created the West Coast Global Warming Initiative, and in 2006, Arizona and New Mexico launched the Southwest Climate Change Initiative. The provincial premiers of British Columbia, Manitoba, Ontario, and Quebec, and the governors of Montana and Utah have also joined the original five western states in committing to tackle climate change at a regional level. The seven U.S. states represent about 20% of the U.S. economy while the four Canadian provinces make up 70% of the Canadian economy (see Brohé et al. 2009). The WCI cap-and-trade program is expected to cover about 90% of the GHG emissions in participating American states and Canadian provinces. While the regional WCI cap will be the sum of all the individual partner caps, the program's overarching goal is to reduce GHG emissions to 15% below 2005 levels by 2020. To do this, the WCI regional goal remains consistent with the state and provincial goals and does not replace the individual goals. The specific individual caps and the combined regional cap for the years 2012 through 2020 are to be set prior to 2012 when the program begins.

Finally, on November 2007 nine Midwestern governors and two Canadian provinces signed on to participate (Iowa, Illinois, Kansas, Manitoba, Michigan, Minnesota, and Wisconsin) or observe (Indiana, Ohio, Ontario, and South Dakota) in the Midwestern Greenhouse Gas Reduction Accord (MGA).

Under MGA, members agreed to establish regional GHG reduction targets, including a long-term target of 60%–80% below current emission levels and a multisector cap-and-trade system to help achieve the targets.

The Chicago Climate Exchange (CCX) is a GHG cap-and-trade system in which a group of nearly 300 North American companies (Motorola Inc., Ford Motor Co., IBM, Intel Corporation, and DuPont, among others) voluntarily agreed to reduce GHG emissions 6% below a baseline period of 1998–2001 by 2010. These companies can comply with annual GHG emission reduction targets through internal reductions, purchase of allowances from other

companies facing emission limitations, or purchase of credits from emission reductions projects that meet specific criteria. The CCX launched trading operations in 2003 with thirteen members. However, the CCX clearly has benefited from the overall direction of climate policy within the U.S. as new regional initiatives began to take shape and as interest in and prospects for an economy-wide federal cap-and-trade scheme grew. In October 2007, CCX claimed that its recent growth in membership had brought more than 540 MtCO₂e below its cap (or about 7%–8% of reported 2005 U.S. GHG emissions). The prospect of U.S. engagement in climate policy also attracted a major new exchange, the New York-based Green Exchange, into the market. Though CCX ceased trading carbon credits at the end of 2010, during the years it was operating some good effects were realized: it helped businesses to prepare for potential regulations of greenhouse gas emissions at the international, federal, or regional levels, and facilitated investment in new technologies.

Emissions Trading in Europe: EU ETS²

European environmental policy has been traditionally dominated by command and control policies and emission taxes. However, the political momentum after the negotiations of the Kyoto Protocol—where emissions trading was included as one of the possible “flexible mechanisms,” along with the Clean Development Mechanism (CDM) and Joint Implementation (JI)—favored the debate about a European trading regime. Partly this was a concession to the U.S., and partly it was born out of frustration with an attempt at EU-wide taxation that failed some years earlier. (The failure that time was due almost entirely to the fact that it was the European Union that was seen as taking over the right to tax, which is zealously defended as a national prerogative by finance ministers and national governments in Europe; also, during the tax debate in the 1990s, climate was not yet perceived as such a serious threat as it is today). In addition, experience with several small experiments in CO₂ trading within Europe—the UK Emission Trading Scheme, the Danish CO₂ trading program, the Dutch offset program, and BP’s trading program—made emissions trading a less foreign concept.

The first detailed discussion of a tradable permit system began in March 2000 with the issuance of the commission’s Green Paper on GHG Emissions Trading. A number of working groups were set up to consider how the system might operate and how feasible it might be. Based on the findings and recommendations of all working groups, a proposal was published in October 2001 by the European Commission suggesting the implementation of a GHG emissions trading system within the European Community to enable certain businesses and industries to trade their allocations of CO₂ emissions. Two years of reviews and amendments followed the release of the proposal, which was finally enacted in October 2003. Finally, January 2005 saw the launch of the

European Union Emission Trading System—the EU ETS, the largest emissions trading scheme in the world and the centerpiece of EU efforts to reduce greenhouse gas emissions.

According to the burden sharing agreement of the Kyoto Protocol, each country was given a reduction target for 2012; the EU ETS established a first warm-up phase from 2005 to 2007 and a compliance phase from 2008 to 2012, which coincides with the Kyoto Protocol compliance period. Phase I was intended to put member states on the path to compliance by addressing the emissions reductions early enough to avoid dramatic reductions in the 2008–2012 period and to gain experience with the new instrument. The ETS is a mandatory scheme within the EU. That is, all member states must be a part of it. For new countries, participation is a precondition to become a member of the EU. In its current state, the EU ETS includes some 12,000 installations in the heat and power generation industry and in selected energy-intensive industrial sectors. These installations represent approximately 45% of EU emissions of CO₂ and 30% of total EU greenhouse emissions.

One of the most interesting features of the EU ETS program is that it is both centralized and decentralized. At one end, the European Commission (the central authority) determines who will participate in the market and tries to fix how many permits will be created, how allowances will be distributed among various emission sources, and the rules for compliance and trading. At the other end, each of the 25 member states is responsible for the allocation of permits within its territory. The number of allowances that will be given in each country is laid out in a National Allocation Plan (NAP). The total cap in the EU trading system is then given by the aggregate of all member state allocation plans. Particularly in the first round of allocations, it seems there might have been some incentive for countries to try to game the system by creating more permits in their NAP.

NAPs must conform to a number of criteria set by the EU—among them, the total number of allowances proposed by each member should be lower than business-as-usual projections, and they must not preclude achieving the target set by the EU burden sharing agreement of the Kyoto Protocol. Nevertheless, the process of setting up the NAPs turned out to be complex, controversial, and sometimes characterized by lobbying and strategic interactions among industry, member states, and the EU Commission. Concerns over a “race to the bottom” among member state allocations were heightened by the fact that not all NAPs were submitted at the same time. For example, the UK NAP was published early and generally considered relatively stringent. Once other member states had published their NAPs and these had proven to be more lax, the UK filed a request to adjust its NAP and increase its allocation volumes. Although the request was disallowed by the Commission, the example illustrates that the allocation process contains elements of strategic behavior by member states. A centralized allocation at the European level, or at least a common decision on the total volumes to be allocated, would

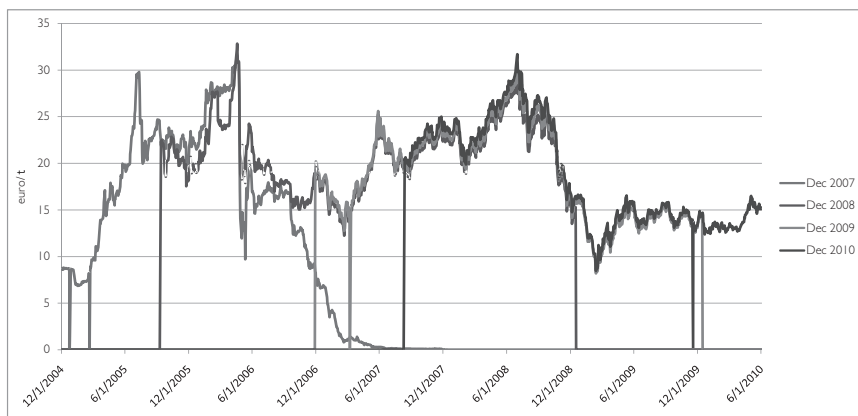


FIGURE 6-1 Price of EU Allowances in the EU ETS.

Source: Point Carbon 2010

mitigate this problem. However, such an approach had little initial support among member states, several of which were still reluctant to endorse the creation of the trading system at all.

Although the European Commission decided to reduce the proposed totals for Phase I in 14 of the 25 NAPs submitted by the member states—representing about 5% of the total cap—early assessments indicated that the allocation was still too generous; the trading system was criticized as too lax even before it was launched (Zetterberg et al. 2004). Nevertheless, the first year of trading saw prices of emission allowances (EUA) higher than many observers had expected, peaking at over 30 €/ton early in 2006 (Figure 6.1). However, the prices fell dramatically in 2006—after the first statistics about verified emissions in 2005 showed that the market was long on allowances. While the obvious explanation of the price break is an adjustment of expectations, EUA prices did not go to zero immediately. Moreover, the prices did not reach near zero levels until a year later as it became increasingly clear that weather and other factors would not create additional demand before the end of the first trial period.

Since very low allowances prices in Phase II (2008–2012) could seriously jeopardize the credibility of the trading scheme, the European Commission repeatedly stated its intention to tighten the cap for the second trading period as member states prepared their NAPs for Phase II. The European Commission completed its review of the NAPs for the now 27 member states by early October 2007. Overall, the reviewed NAPs were cut by 10.4% below the caps originally proposed by member states, leading to a maximum of 2.1 billion tons. This corresponds to a cut of 6.0% below 2005 verified emissions.

The April 2008 release of 2007 verified emissions data was therefore eagerly awaited by market actors and observers since it was considered relevant for the

analysis of estimated shortfall in the Phase II allowance market (2008–2012). The numbers showed that despite a mild winter, emissions had continued to rise within the EU ETS perimeter. Economic growth in the region had been higher than many analysts had expected, and EU ETS emissions in 2007 had grown by an average of 1% per year since 2005—with more vigorous growth in the Eastern member states. This caused some analysts to revise their forecasts slightly upward for the likely shortfall in Phase II, and eventually for their projections of EUA prices in Phase II. However, most analysts agree that the broad international negotiations on post-2012 commitments, as well as the definitive contours of Phase III, will ultimately be decisive for the development of the EU ETS market.

In the road ahead to Phase III, the EU has committed to reducing its overall emissions to at least 20% below 1990 levels by 2020 and is prepared to scale up this reduction to as much as 30% under a new global climate change agreement provided other developed countries make comparable efforts. It also set itself the target of increasing the share of renewable energy in overall EU consumption to 20% by 2020 (including a 10% target for biofuels in vehicle fuel) and a 20% increase in energy efficiency. The package also seeks to promote the development and safe use of carbon capture and storage (CCS). Strengthening and expanding the EU ETS is central to the EU's strategy. Emissions from the sectors covered by the system will be cut by 21% by 2020 compared with levels in 2005, and free allocation of emission allowances will be completely eliminated by 2020. Sectors able to pass along costs (for example, in the power sector) will face full auctioning by 2013, while free allocation will be progressively phased out (from 80% in 2013 to 0% in 2020) for those sectors exposed to international competition.

So far, allowances have mostly been given away free, and allocation has been based on historic figures with some small amount to be auctioned (less than 5% for the first phase and less than 10% for the second phase). A great deal of rent-seeking has taken place in efforts to modify the various allocation principles. It has become common to refer to all free allocation based on some historic data as grandfathering (GF), although true GF implies the creation of rights based on past emissions that are never subsequently changed. EU practice has departed from this ideal in several ways: First, allocation was often in proportion to historic production and some administratively decided ("best practice") level of emission intensity, sometimes referred to as benchmarking. In most countries, we furthermore could see that firms exiting the market (for instance by plant closures) lose their permits and new entrants gain permits. Although both allocation for new entrants and the "use-or-lose" rule for closures may appear to have some common sense, they may introduce a relationship between output and allocation, fostering strategic action by firms in the permit trading scheme. For example, confiscating allowances after facilities close creates a subsidy for continued operation of older facilities and therefore a disincentive to build new and cleaner facilities (see further Müller and Sterner 2008).

Grandfathering has been perceived as unfair by new firms since it implies that “dirty” firms, which get more generous allocations, are favored at the expense of “cleaner” ones. In the worst of cases, firms were actually punished for having taken proactive behavior to reduce their emissions. In addition, the allocation system may have created perverse incentives for firms to engage in potentially costly rent-seeking behavior in order to gain a more generous allocation. Referring to high marginal costs for abatement has been a common strategy used by some industry sectors in order to receive a high allocation in the EU ETS. For example, the steel and cement industries have been particularly active in raising their concerns over the additional costs that auctioning allowances would have on them. Both individual companies and industrial associations have argued that auctioning would be very detrimental to them in terms of the international competition that they face from firms outside of the EU ETS. Most of the potential pitfalls associated with free allocation were known ahead of the start of the EU ETS. However, considering the lobbying power and economic importance of these industries in Europe, it would have been politically very difficult to introduce full auctioning in the first phase of the EU ETS.

In addition to partly decentralizing the choice of cap and allocation, member states were free to choose whether and how to allocate banking between 2005–2007 and 2008–2012 and to establish domestic compliance procedures—monitoring, reporting, and verification—that are increasingly important as the EU is expanded to countries with weaker institutions. In practice, most member states ruled out the banking option, probably because any banked allowances would have been subtracted from the country’s 2008–2012 Kyoto targets. This reduction in the pool of free allowances in the future is actually a clear disincentive to banking.

Early analyses have shown that there could be several important differences in the monitoring and enforcement procedures among the member states. Discretion on the interpretation of monitoring guidelines may undermine some of the consistency that is necessary for an effective regime. However, initial variation in the application of guidelines will diminish over time as the European Commission is asserting a more centralized role and taking measures to harmonize procedures. Perhaps more serious are the broader differences in legal systems, enforcement cultures, and administrative capabilities among the member states, which could create unfair competitive advantages for firms in member states with weaker enforcement regimes. Clearly, there are no quick solutions to these issues, and they just mirror the broad variation in regulatory institutions and practices throughout the EU.

Although it may take time to reach a truly definitive assessment of the effects of the EU ETS on producers’ and consumers’ choices, economic activity, trade patterns, and GHG reductions, the EU ETS has undoubtedly succeeded in imposing a price—by far the most significant accomplishment in climate policy to date (see, for instance, Ellerman et al. 2010). Emitters of CO₂ in the EU now face price signals that reflect the fact that Earth’s capacity to absorb

greenhouse gases is limited. The price signal goes beyond the EU ETS; it also provides a basis for evaluating initiatives in the CDM, JI markets, other trading under the Kyoto Protocol and beyond, and emerging opportunities for CCS. On the other hand, the model of governance in the EU ETS has broken new ground in the experience with emissions trading regimes across multiple jurisdictions. It is providing new evidence on how variations in the institutional capability to implement emissions trading can be encompassed in a single trading program; this evidence is particularly relevant if we think about how a global trading regime might evolve. It is possible—indeed, likely and desirable—that the EU ETS will be either developed or superseded by better instruments. Meanwhile its very powerful precedent has raised consciousness both concerning permit trading and climate change in the world. Considering the scale and urgency of the climate challenge, the EU ETS should be judged first and foremost on the merit of having been a pioneer activity.

Finally, it is interesting to reflect on the apparent contradiction that the EU ETS was carried forward by the central institutions of the emerging European Union, while the most effective climate policies in the U.S. appear to come from the decentralized level of local, regional, and business circles. Clearly, climate policy is an issue of the future, and in Europe it provided the central authorities an opportunity to be more visionary; by seizing the initiative they have also advanced the cause of European integration. Initially they ceded considerable power to the nation-states over allocation, but when the results were incoherent and marked by rent-seeking this provided a probably welcome opportunity to move toward greater central control. In the U.S. it would seem as if the opposite happened (although this might well turn out to be temporary). The vacuum left by the Bush administration in this area created room for state initiatives, but it would not be surprising if this in turn leads the federal institutions to regain the initiative.

Green and White Certificates in Europe

To arrest climate change, a transition to a low-carbon economy must take place quite rapidly—within a century at most (Jacobsson and Lauber 2006). Therefore, the development and diffusion of renewable energy sources has become a major mitigation strategy. For instance, under the Kyoto Protocol, the European Union has agreed to reduce greenhouse gas emissions by 8% between 2008 and 2012 (relative to 1990 levels). As a result of this agreement and recent price increases on the oil market, The European Commission has formulated the Community aim to supply 20% of electricity consumption in 2020 with renewable energy sources. New EU energy and environmental policies have been developed in several European countries to foster market penetration of renewable energy sources. These include an array of policies, including carbon taxation, subsidies for renewables, and feed-in tariffs. One of the new strategies that has the characteristics of property creation is the

establishment of a market for tradable green certificates (TGCs). Under this scheme an obligation to supply or consume a certain quota of renewable energy (or the equivalent amount of green certificates) is imposed on one category of electricity system operators (generators, producers, wholesalers, retailers, or consumers). On a settlement date, the operators must submit the required number of certificates to demonstrate compliance.

Certificates can be obtained in several ways. Operators can generate their own renewable energy and so become producers of certificates. They can purchase electricity and associated certificates from eligible renewable electricity generators. Or they may purchase certificates that are traded independent from the power itself. By principle, green certificates conform to the same basic structure under all these alternatives: a tradable instrument that represents the renewable energy attribute of physical electricity and that confers property rights to the holder. On the other hand, because of supply-side competition, such schemes should achieve the intended quota at the minimal generation cost.

In addition to the mandatory obligations, voluntary markets for green certificates have emerged in response to the demands from energy-conscious electricity consumers that are concerned about global warming and want to influence their own fuel mix. Some electricity suppliers differentiate themselves from their competitors by providing “green” electricity—typically at premium prices. For example, consumers in the United Kingdom can purchase 100% green energy at an average rate of £0.13 kWh plus a daily fixed charge of £0.13, as opposed to the average electricity price of £0.126 (Association of Issuing Bodies 2010). However, in order to prove the renewable origin of the electricity, electricity producers must provide a guarantee of origin that enables the consumer to claim the specific source of the consumed electricity.

During the past few years, interest in green certificates has increased. Markets have been established in a number of EU member states, including the UK, the Netherlands, Italy, Belgium, Poland, Romania, Sweden, and Austria. Indeed, the amount of certificates issued has increased over the last decade. Since 2001, more than 779 million have been issued, of which 485 million have been used to guarantee to consumers the origin of the renewable energy they have purchased. In 2009, more than 209 million certificates were issued, and 151 million were used to disclose the source of energy to consumers (Association of Issuing Bodies 2010).

Nevertheless, schemes differ considerably in design, particularly when it comes to the classification of “green” technologies. Most schemes include only innovative green technologies which would not be commercially viable without economic support, and thus exclude some renewable sources such as large-scale hydro power and some pre-existing other renewable installations. Conversely, some schemes include energy sources that are based on “conventional knowledge” and that are not strictly “renewable,” such as fossil fuel-fired combined heat and power (CHP) generation. For example, Sweden awards

BOX 6-2 The Swedish System for Green Certificates

According to the 2009/28/EC EU directive, Sweden must reach a target of 49% of the total energy supply from renewable energy sources by 2020; in this context, the tradable electricity certificate system has been recognized by the Swedish Parliament as the most important mechanism to achieve such a goal. The system was introduced in May 2003 with the explicit goal of increasing the share of renewable electricity production by 10TWh by 2010 (Nilsson and Sundqvist 2007). Nevertheless, on January 1, 2007, the target was revised upwards, calling for an increase of 17 TWh by 2016 in production of electricity from renewable sources relative to the 2002 level. In June 2009, Parliament confirmed a further, higher target for the system, of achieving an increase on the lever of 25 TWh by 2020.

All electricity suppliers and certain electricity users (electricity-intensive companies) are required to purchase certificates corresponding to a certain proportion (quota) of their electricity sales or electricity use; this quota is increased yearly, thus generating an increasing demand for the certificates. The certificates are not restricted to a certain quota period, thus banking between periods is allowed. Noncompliance results in a penalty fee equal to 1.5 times the price of the certificates.

The cost of purchasing the required certificate is forwarded to the end-users of electricity. Therefore, it increases the retail price for consumers. Producers of “green” electricity are allocated one certificate for each MWh electricity produced from eligible renewable sources, such as wind power, solar energy, wave energy, geothermal energy, biofuels, peat and hydro power (Swedish Energy Agency 2009).

The total number of plants approved for allocation of certificates in 2008 was 2,254, of which more than 50% corresponds to hydro power plants, although to date the largest fraction of certificates has been produced through the use of biofuels. A small number of solar energy plants were approved for allocation of certificates, but they produced few MWh of electricity. No wave energy or geothermal energy plants have so far been submitted for approval and inclusion in the system (Swedish Energy Agency 2009).

Trading, both spot and forward, on the electricity certificate market occurs through bilateral agreements, directly between producers and those end-users having quota obligations, and through contract purchases intermediated by brokerage houses. Ever since the inception of the program, there has been a surplus of certificates on the market. In 2008, for example, this surplus amounted to approximately 5 million electricity certificates (Swedish Energy Agency 2009). So far, the price levels on the certificate market have exceeded expectations. For example, in 2004, the actual prices ranged from 210–250 SKr per certificate when expected prices were 50–160 SKr. Nevertheless, calculations by the Swedish Energy Agency indicate that a considerable amount of the money never reaches the renewable electricity producers; instead, it is taken by the high transaction costs of the system and rent-seeking behavior by the retailers (Nilsson and Sundqvist 2007).

certificates for electricity produced from biomass which, it is often asserted, would be competitive anyway. Thus windpower gets relatively less support and virtually none goes to truly innovative technologies such as solar. Even peat-based generation is eligible (in CHP plants); therefore, such sources become competitive at comparatively low TGC prices (Unger and Ahlgren 2005).

Due to the differences across national schemes, green certificates are most often not tradable internationally, which constitutes a primary impediment for further efficiency gains as well as increased liquidity and reduced risks of market power. However, significant efforts have been undertaken to encourage the standardization of green certificates across countries, aiming for the development of an integrated European market. The issues are similar to the “linking” of regional carbon trading schemes: On the one hand promising opportunities for trading appear, but on the other many of the political economy issues that originally prevented an international agreement on permit caps and allocation principles (thus leading to regional schemes) may resurface when schemes become linked.

Following the green certificates experience, attempts are being made at the EU level to apply a similar concept to enhance energy efficiency through the use of “White Certificates,” also referred to as Energy Efficiency Titles (EET) or Energy Efficiency Certificates (EEC). The basic idea is that energy suppliers or distributors must fulfill specific energy saving targets by implementing energy efficiency measures toward their clients within a specific time frame. As in the case of green certificates, the White Certificates are a market-based mechanism that allow operators to decide whether to implement energy efficiency measures themselves or to buy certificates in order to comply with binding administrative targets (in this sense the instrument also builds on pre-existing regulations). Variations of this instrument are currently in place in Italy and the UK, about to be introduced in France, and under serious consideration in Denmark and the Netherlands (Oikonomou and Mundaca 2008; Sorrell et al. 2009). Currently, White Certificates are mainly targeting power and gas suppliers by imposing energy efficiency obligations, but they also include nonenergy intensive sectors such as the commercial and residential sectors. The projects that lead to the generation of certificates can be very generic, such as the installation of energy efficient light bulbs, or specific, requiring the use of specific technologies in production processes. Estimates indicate that their use could increase rapidly within Europe; they can potentially cover about half of the total natural gas consumption and 70% of the total electricity consumption in the EU-15 countries (Austria, Belgium, Denmark, Finland, France, Germany, Greece, Ireland, Italy, Luxembourg, the Netherlands, Portugal, Spain, Sweden, and the United Kingdom) (Farinelli et al. 2005).

The issue of “additionality” is quite relevant when it comes to “White Certificates,” since they are supposed to encourage energy efficiency measures that would not have occurred under a business-as-usual scenario. As with the CDM, this issue is still debatable. Nevertheless, some country-specific criteria used to determine additionality include measures that exceed what current law and building regulations require as well as spontaneous market trends and legislative requirements (Mundaca 2008).

Although the introduction of such schemes might have some positive effects in terms of reducing the cost of CO₂-emission reductions, their indirect

effects are difficult to track since energy markets are integrated, and the design of instruments in one country interplays with markets and instruments in other countries. On the other side, there are also strong interactions between these schemes and the EU ETS. For example, Unger and Ahlgren (2005) have shown that the introduction of green certificates reduces wholesale electricity and emission permit prices within the EU ETS, with the impact on the latter being rather pronounced. A similar effect is caused by the use of white certificates (Sorrell et al. 2009). Most important, it is not clear that these approaches will make any contribution to reducing EU or global carbon emission unless the EU ETS cap is tightened. On the other hand, if the promotion of renewable electricity is to reflect higher ambitions than the current EU ETS cap, support schemes to ensure long-term development for certain new promising technologies is likely to be necessary. Furthermore, green and white certificates might be insufficient and should be complemented with traditional governmental support programs if they are to promote the development of truly innovative technologies.

Other Emission Trading Programs

The concept of permit trading is spreading. Countries such as Chile and Poland (see Chapter 21) introduced trading schemes during the 1990s to reduce air pollution (Stavins and Zylicz 1995) while oil companies such as BP, Shell, or even *Petróleos Mexicanos* (PEMEX) introduced voluntary internal GHG emissions trading schemes starting already in the late 1990s. In 1994, trading experiments were conducted in five cities of China (Baotou, Kaiyuan, Liuzhou, Pingdingshan and Guiyang) to control SO₂ emissions. The pilot programs took different forms, including allowance transfers within an enterprise, investments in nonpoint-source pollution control and transfers from sources with surplus to new or existing sources with insufficient allowances. However, lacking a legal foundation for emission trading, the programs did not succeed in establishing themselves across the country nor in enduring over time. Nevertheless, the interest in the use of trading has grown over time as some of the conditions necessary for emission trading have developed (Yang et al. 2003).

In the international arena, many donors and advisors are promoting the use of market-based instruments as the key to more effective environmental protection in the developing world. The least-cost property of market-based instruments is particularly appealing in these countries, where the competition to meet social needs is great and the income level is relatively low (see Chapter 21). However, a main issue affecting the designing of tradable permit programs in the developing world is the lack of legal infrastructure, as well as monitoring and enforcement capabilities that would be required to ensure compliance. While this constraint is likely to remain over time in some developing countries, the application of emissions trading will nevertheless still be

appropriate for others. Clearly, as the national circumstances in developing countries vary widely, so will the individual capacity of developing countries to apply such schemes. One of the main economic and developmental challenges in many developing countries is the lack of suitable tax bases: many countries have unsustainable levels of budget deficit or foreign aid revenue shares. Since income tax is hard to organize, and trade and other taxes are distortive, taxation of externalities may be an attractive option.

Water Trading

Water trading is as an allocative mechanism in which entitlements of water quotas can be transferred between users, intra-valley (within the same water basin) or inter-valley (between water basins). Water is thus reallocated to users with higher values and incentivizes the holder of the right to conserve water; the potential purchasers are also incentivized to use efficient technologies to reduce water consumption and water losses in order to reduce costs of purchase.

Water trading can be done either through water right trades, in which the object of trade is the right to be allocated a certain volume of water from the managing agency, especially for irrigation purposes, either through effective transfer or water volumes. Apart from the traditional water transfers in the form of sales or leases, numerous innovative water transfer arrangements exist: callable water-use options that allow a city to lease water under specified drought conditions; water leasebacks, in which a municipality purchases the right and then leases the water back to agricultural users in nondrought years; water right priority exchanges that shift the drought-year allocation queue; or banking water by artificially storing (often imported) water in underground aquifers during wet years to be pumped out during dry years (Chong and Sunding 2006).

Water trading is of course used to ensure sufficient irrigation resources, but urban water trading schemes can also be implemented, in which urban households and industrial water users buy and sell water entitlements and water allocations. Nevertheless, transaction costs might be an important impediment to fully reaping the benefits of trade. The costs of identifying potential trades, recording and enforcing changes in water intakes, and conveyance infrastructure may be very large in comparison to the value of water. Hence, a trading system would be justified only in areas confronted with severe enough water shortages. The institutional costs associated with establishing and enforcing the rights might also be very high, with both public and private institutions such as water user associations being required to operate at the level of ditch, canal, and river basin. The common perception that “everyone” has an inherent right to some water may create social disapproval of full privatization and prevent trading by not respecting private property rights over water.

On the other hand, large hydrologic effects can occur as a consequence of inter-basin trading if trading is permitted without a complete understanding of the water system. If the trading scheme does not ensure minimum water flows at the lower sections of the rivers, aquifers can be depleted, increasing water pollution and changing ecosystems. Water trading is also contested on the grounds that it reallocates water resources from productive agriculture to urban uses, though the empirical evidence is mixed in this direction; however, operational rules are enforced in some regions to explicitly prevent large agriculture-to-urban transfers. Distributional asymmetries can also appear due to water trading, with large economic losses suffered by the area-of-origin of the water. The most well-known example is the water trading that took place during 1905–1935, when Los Angeles secretly purchased water rights from an agricultural area in Owens Valley: the sellers were paid a higher price than the agricultural value of their land, but much smaller than the economic value to the buyer district, raising equity and fairness issues. Negative local impacts on the area of origin have been documented, especially over the short term, when the local economy has not reinvested the proceeds of the water rights sale.

In spite of these pitfalls, water trading has been used successfully around the world: in southeast Australia since 1980 in the Murray-Darling Basin (MDB); in wide regions of the U.S. such as the Colorado-Big Thompson area; in Latin-American countries like Chile and Mexico; and African countries like South Africa. In Australia, the most important water rights transfer system is within the MDB, its most significant river system and home to the nation's three longest rivers—the Darling (2,740 km), Murray (2,530 km), and Murrumbidgee (1,690 km). The MDB covers 14% of Australia's total area and is the country's most important agricultural region and dependent on the irrigation system. Around 70% of all water used for agriculture in Australia is used by irrigation in the MDB.

The water market allows for two types of trades: permanent trade and temporary trade. Permanent trade of water entitlements is the transfer of the ongoing right to access water for the term of the right. Temporary trade of seasonal water allocations is the transfer of some or all of the water allocated in accordance with the entitlement for the current irrigation season or for an agreed period of time.

Water trading was initially confined to trades within irrigation systems, but as larger gains stem from inter-valley transfers, trading rules have been modified to permit inter-valley and more recently interstate trade. In recent years, Australian governments have been working together to reduce the differences in water entitlements (caps) to prepare for the introduction of increased interstate water trading.

The extent of actual trades on a permanent basis has been relatively modest, although temporary trade has grown significantly. Public water exchanges have been created to facilitate exchange in seasonal water allocations by providing a public notice board and a clearinghouse. The largest of

these public clearinghouses is Watermove, which covers trade in the large irrigation districts on the Victorian side of the Murray and the Goulburn Valley. The persons eligible to trade are the individuals or farms supplied by the area's corresponding water authorities, and the prices formed on the exchange are largely dependent on the seasonal allocation, the water use in the current season compared with previous seasons, gross margins or likely returns that potential traders could receive from using the water, cost of alternatives (e.g. dairy farmers buying feed), volumes available for sale, the tariff structure for water, and others (Watermove Exchange 2010).

Despite all its free market characteristics, the water trading system implemented in the MDB irrigation districts has been constrained by administrative and regulatory arrangements, justified by potential market failures or negative externalities which would occur otherwise. The justifications include: (1) hydrological limitations to water movement due to channel capacity constraint; (2) environmental impacts of changing the current patterns of the supply and use of water; (3) stranded assets where an irrigation authority is faced with large fixed infrastructure costs and a declining customer base; and (4) social and economic adjustment costs associated with water being exported from particular districts (Qureshi et al. 2009). A restriction in place refers to the water authorities in Victoria, which may refuse permanent trade out of an area if annual net transfers exceed 4% of water rights in that area, and also may ban interstate temporary trade following the end of the irrigation season.

Externalities which may arise from inter-valley water trading are dealt with by using an exchange rate system among states. For example, trades from upstream diverters from New South Wales to Victoria and from Victoria to South Australia have an exchange rate of 1.0 (100% of the entitlement can be transferred downstream), but transfers from South Australia to the upstream states of Victoria and New South Wales have an exchange rate of 0.9 so that only 90% of the entitlement can be transferred. The application of an exchange rate enables the volume and reliability characteristics of the water entitlement to be converted from those of the seller's state to those of the buyer's state, including accounting for losses incurred in delivering the water (Qureshi et al. 2009).

The gains from the MDB trading system have been estimated at AUD\$88 million, with MDB-wide net returns ranging from AUD\$2.5 billion to AUD\$2.6 billion depending on the simulation scenario (Qureshi et al. 2009). Other studies have estimated the gains from free trade in the MDB to AUD\$555 million in dry years and AUD\$201 million in wet years. A computable general equilibrium model used by Peterson et al. (2005) which includes impacts on other industries and sectors estimates the gains of trade in the MDB of AUD\$143 million in dry years and AUD\$44 million in wet years, with a mean value of AUD\$94 million.

Trading Programs for Other Resources

Another major and generally successful set of tradable permit programs is the ITQ program for fisheries (see Costello and Deacon 2008 and Chapter 25). Many of these programs have dramatically reduced excessive fishing and thus restored profits and saved fish stocks. Despite this success, the process has not been without problems. Fisheries are a valuable resource, and the concentration of shares in the hands of a relative minority has created considerable social tension in some fishing communities. Also, some problems remain, such as the discarding of juvenile fish. Still, the overall picture is positive, and one of the contributing innovations is that the quotas are not for fixed harvests of fish but for fixed percentages of a total allowable catch. The total allowable catch, in turn, can be changed at short notice in response to variations in the stock.

Other examples of tradable permit programs in natural resources management include grazing rights and transferable development rights (TDRs) for land planning. TDRs allow land planners to overcome many of the shortcomings associated with traditional zoning practices. A TDR program restricts development in one zone, for instance, to allow for the creation of a park. In exchange, the landowner is given the right to transfer a “development right” to another zone where development is permitted with the help of TDRs purchased from the first zone. To create a green zone around a city usually entails the problem and cost of expropriating properties from landowners. Using TDRs, these landowners are partially compensated, and a large group of landowners shares the burden. However, numerous legal issues still surround this instrument (Miller 1999). Among the latest innovations in trading are schemes for river or wetland restoration (see, for instance, Palmer and Filoso 2009).

Supplemental Reading

Emissions Trading Programs (U.S.)

Brohé et al. 2009
Burtraw and Palmer 2003
Burtraw and Szambelan 2009
Hahn 1989
Krupnick 1986
Portney 1990
Stavins 2008
Tietenberg 1985, 1990

Emissions Trading Programs (Non-U.S.)

Convery 2009
Ellerman and Buchner 2007
Ellerman et al. 2010

Trading Programs for Water and Other Resources

Costello and Deacon 2008
Chong and Sunding 2006
Palmer and Filoso 2009

Notes

- 1 There may still be some difference between grandfathering and auctioning due to wealth effects, and there could be some effects on entry and exit—and thus on industry size and structure. The major difference is likely to be in the political feasibility of a given permit scheme. If there is no difference, then the only arguments in favor of grandfathering would appear to be either a “prior appropriation” concept of rights or simply the bargaining power of the polluters.
- 2 Many thanks to Markus Wråke in this section.

Subsidies, Deposit–Refund Schemes, and Refunded Emissions Payments

Chapter 5 was about various forms of taxes or charges. The “other kind” of price instrument is often said to be a subsidy. Although these two instruments may seem to be worlds apart to a layperson, economists tend to think of subsidies as similar to taxes. They can be partly analyzed as negative taxes, and some reflection shows that they are indeed similar in many superficial respects, but there are also crucial differences between taxes and subsidies.

At a deeper level, the main differences between taxes and subsidies are their implications for the questions of ownership and rights to nature. Also, more combinations of and hybrids between taxes and subsidies exist than might at first be apparent. They can be very interesting from a policy perspective because they allow decisionmakers to keep some of the positive aspects of an instrument while avoiding some of the negative ones.

Subsidies and Subsidy Removal

There are many forms of subsidies, from tax expenditures (see Chapter 5) to more classical direct, budget-financed payments in support of certain activities (or people). Subsidies may apply to payment for certain “services,” prices for certain inputs or technology, loans, or access to credit markets.

As shown in Box 7-1, subsidies work similarly to taxes, but their properties with respect to the number of firms in an industry are different. Subsidies lack the output substitution effect of taxes. Not only are the price and thus the output effect missing, but at least some subsidies (like the kind analyzed in Box 7-1) create a perverse or opposite effect because they tend to encourage the entry (or delay the exit) of polluting firms, resulting in too many firms and too much production and pollution—possibly even more pollution than when unregulated (Baumol and Oates 1988) (see Chapter 11).

The perverse output effect of subsidies can be reduced by careful design (this is the other alternative mentioned in Box 7-1). Instead of a payment for

BOX 7-1 Similarities and Differences between Subsidies and Taxes

A subsidy can be either a direct (partial) repayment of abatement costs or a fixed payment per unit of emissions reduction. In the latter case, the subsidy can be seen as a kind of negative tax. Suppose that the polluting firm (i) has an initial emissions level of e_{i0} and is paid a subsidy T for each unit of emissions abated beyond the initial level. The polluter then receives a subsidy $T(e_{i0} - e_i) = Te_{i0} - Te_i$, which can be seen as almost identical to the tax case in Equation 5-1 (see Chapter 5) if T is set equal to the Pigovian tax.

In fact, the subsidy is a combination of a variable tax (Te_i) and a fixed subsidy (Te_{i0}). Because the subsidy is fixed, this component does not affect the first-order conditions for optimality of a representative firm, which are still the same as Conditions 5-2 and 5-3 (see Chapter 5). Thus, this firm will have the same incentives for abatement, which shows that there is indeed some similarity between a tax and a subsidy. However, differences stem from the fixed subsidy. It lowers the total and average cost to the firm, making it lower under subsidies than under taxes. A marginal firm that would exit the market (produce zero or go bankrupt) under taxes might survive under subsidies. Consequently, the whole industry would tend to have too many firms and produce too much output.

In terms of allocative efficiency, this scenario is a loss (although one can imagine settings where it may be seen as an advantage). In the long run, entry might even be encouraged, and the number of firms in the industry would differ as between a subsidy scheme and a tax regime. On the other hand, if the subsidy covers only abatement costs, then there is no such incentive for entry.

each unit of abatement compared with a baseline, the subsidy is typically a (partial) repayment of verified abatement costs (often fixed capital costs for a filter, catalytic converter, or some other item). The subsidy instrument does not fulfill the “polluter pays” principle, but (partly for that reason) it is popular with polluters and in some cases may be feasible from a practical standpoint, particularly when other instruments are infeasible—e.g., when no polluter can be identified, as is the case with some historic cases of pollution, particularly in developing or formerly planned economies. If the polluting company is nonexistent, bankrupt, or unidentifiable (such as after some types of oil spill), then the public sector may have little choice but to finance cleanup with public funds—or subsidize other companies to do the cleanup. The same applies if, for other reasons, the public sector has insufficient power. However, making an industry collectively responsible for certain kinds of cleanup may lead to conflicts.

Other situations in which subsidies might be warranted would be when the polluter expressly owns the property rights to a certain resource that society wants, or in financing research or development that has strong public good characteristics. Such research is typically underfinanced because firms cannot profit from it. In many countries, the credit market is described as “imperfect” and skeptical toward loans for the financing of investments in pure abatement. This lack of research provision is also an instance when subsidies (or

public provision) may be warranted. The most practical argument against subsidies is that they are too expensive as a policy instrument—especially in developing countries, where the opportunity cost of public funds is high. In such countries, taxes are particularly hard to collect, and many pressing needs (such as primary schooling and health) need to be satisfied.

In reality, the most relevant issue for the environment is not subsidies for abatement but the prevalence of perverse subsidies for pollution. Inappropriate subsidies promote rather than prevent wasteful and environmentally destructive behavior. Well-known examples include large subsidies for energy use in many countries, particularly oil-exporting countries and the formerly planned economies (Kosmo 1987). The formerly planned economies subsidized the domestic consumption of not only energy but all natural resources (Bluffstone and Larson 1997). In the fishing industry, harmful subsidies (that increase rather than decrease effort) are common, and enormous subsidies for forest destruction in Brazil have been documented in recent decades (Binswanger 1991; see also Chapter 27). Perverse subsidies are so common that “subsidy removal” is often classified as an environmental policy instrument in itself.

It is somewhat ironic that the undoing of one bad policy should be classified as a new policy instrument. However, the removal of subsidies is politically complicated, because subsidies become intertwined with vested interests. In fact, the value of subsidies (or of tax expenditures on goods and services that are implicitly subsidized because they ought to be subject to Pigovian taxation) typically is capitalized in property values. If an individual buys a house with electric heating in a cold climate, then the dependability of the heating system is one of the most important attributes of the house. If the government changes the value of this attribute after the house is bought by abolishing nuclear power or taxing fossil power, then the value of the house may plummet in expectation of future energy bills. Properties acquired just before a policy change can suffer particularly serious losses in value. In general, the political economy of subsidies is based on the fact that those who benefit are often few and well organized, whereas those who stand to lose are many and have a small, diffuse interest (see Chapter 13).

Deposit–Refund, Tax–Subsidy, and Other Two-Part Tariff Systems

In addition to the basic policy instrument types already mentioned, several more complex instruments exist, many of which are combinations of the other instruments. In fact, combined instruments is a fertile area. Quantity restrictions, for instance, can be combined with a (small) subsidy for over-compliance or with (high) fees for pollution emitted above a certain level. Such combinations may serve as a “safety valve” if regulators are not sure of the optimal pollution level (see Burtraw et al. 2009 and Chapter 10). They also may provide a way to collect information on abatement costs.

BOX 7-2 The Technical Operation of Deposit–Refund Systems

If the output (q) in our model is identical to the pollution (e) to be abated, as in the case of cadmium batteries, and abatement (a) entails the collection of used product for recycling, then an equation such as Equation 5-1 can model a simple damage function for firm i : $e_i(q_i, a_i) = q_i - a_i$. The first-order conditions are then simply

$$P = c'_q + T \quad (7-1)$$

$$c'_a = T \quad (7-2)$$

where P is product price, c is production costs, and T is the value of the refund. The damage cost of inappropriate disposal is included in the price, and there is an incentive for “abatement” (collection and return) up to the value of the refund.

Some combinations are so well established that they are considered policy instruments in their own right. One such instrument is the deposit–refund system, which encompasses a charge on some particular item and a subsidy for its return. This instrument can be used to encourage environmentally appropriate recycling. Assuming disposal is inappropriate for ecological reasons, the deposit–refund combination may be categorized as a tax expenditure or a presumptive tax on inappropriate disposal. The polluters (i.e., those who do not return the item) pay a charge, whereas those who return the item collect a refund and thus pay nothing (see Box 7-2). The distinguishing feature of the deposit–refund system is that it has a clever disclosure mechanism: the refund is paid when the potential polluter demonstrates compliance by returning the item that carries the refund, thus making the monitoring of illegal disposal unnecessary. Recent developments in deposit–refund systems show that the deposit and the refund need not be the same amount. In fact, a revenue-neutral construction would have higher refunds in general if the rate of return were incomplete (Mrozek 2000).¹

Usually, deposit–refund systems are used for certain final outputs (beverage cans and bottles are the classic examples), and abating environmental pollution has been far from the only (or even main) motivation. However, the concept is spreading, and Sweden has instituted a deposit–refund scheme on scrap vehicles to prevent such vehicles from being left to rust in the woods. It is conceivable that a similar instrument could be used for other polluting inputs, such as cadmium or mercury, but practical complications might arise (e.g., rogue imports from other countries for the sole purpose of collecting refunds).

Wastes generally can be disposed of by appropriate (municipal) waste management or by irresponsible “dumping.” If there were no monitoring problems or irresponsible dumping, then the appropriate “first-best” instrument would be a Pigovian charge on dumping. If this charge is not feasible (because monitoring is impossible), then the charge must be applied to all

goods at purchase and thus becomes a general consumption tax; if the good is properly returned, then the tax is refunded (Fullerton and Kinnaman 1995). The new waste disposal “charge” is thus composed of two parts: a standard waste disposal charge (to cover the cost of municipal disposal) and a refund. This amount is equal to the difference in externality between dumping waste illegally and disposing of it through the proper systems. The latter is assumed to be better from an environmental viewpoint, and thus this whole term is negative. Whether the amount is large enough to outweigh the first term is an empirical matter, but if it is, the optimal fee to the consumer for ordinary waste management might become zero, or even negative. Thus, a type of deposit–refund system has been created in which a deposit is paid on all purchases and later refunded in connection with recycling or (municipal) waste collection, which also is subsidized (applications are discussed in Chapters 17, 19, and 24).

Even though deposit–refund systems have been used mainly for waste management and recycling, the same concept can be applied in other two-tiered instruments by combining a general charge with a subsidy to encourage an environmentally beneficial behavior. In cases of asymmetric information (when the regulator has difficulty in collecting information about pollution), it may be effective for the regulator to design contracts or menus of instruments from which the agents can choose. For instance, a high tax on pollution will discourage firms with high abatement costs from disclosing their pollution levels. However, if high taxes are combined with subsidies for truthful disclosure (e.g., for measurement equipment and technical help, or certain types of abatement investments), then the combination can be made attractive even to high-level polluters.

An especially interesting two-part instrument is the tax–subsidy scheme, in which polluters are assigned property rights to a “baseline” emissions level set by regulators (Pezzey 1992; Farrow 1995). Polluters pay a charge per unit of emissions above the baseline and receive a subsidy per emission “saved” whenever emissions are below the baseline. New firms have no property rights and must pay for their emissions. This instrument has some attractive features. It provides a clear link between the extent of a polluter’s property rights to the environment and the size of the baseline on the one hand, and hence, the extent to which society or the polluters pay the scarcity rent on the other hand. It also is general enough to encompass both subsidies and Pigovian taxes as special cases. If the baseline is zero, then the instrument is a tax; if the baseline is equal to, say, current (unregulated) emissions, then the instrument becomes a generous subsidy.

Refunded Emissions Payments

Another two-part instrument with some interesting properties is the refunded emissions payment (REP) (Stern and Höglund 2000). An REP is a charge, the

BOX 7-3 The Refunded Emissions Payment

Each company (i) seeks to maximize profit as in

$$\max Pq_i - c_i(q_i, a_i) - Te_i(q_i, a_i) + q_i/(\sum q_i)T[\sum e_i(q_i, a_i)] \quad (7-3)$$

where q is output, c is production costs, a is abatement, Te_i is the charge, and the last term is the refund. The first-order conditions for a firm with positive output and abatement are

$$P = c'_q + Te'_q(1 - \sigma_i) - T(E/Q)(1 - \sigma_i) \quad (7-4)$$

$$c'_a = -Te'_a(1 - \sigma_i) \quad (7-5)$$

where $\sigma_i = q_i/Q$ is the firm's market share, Q (gross output) $= \sum q_i$, and E (total emissions) $= \sum e_i(q_i, a_i)$.

The social planner is assumed to maximize welfare (W , see Conditions 4-2 and 4-3 in Chapter 4). Comparison with these conditions and with Conditions 5-2 and 5-3 (Chapter 5) for Pigovian taxes shows two differences between refunded emissions payments and a pure tax instrument. First is a factor $(1 - \sigma_i)$ in Condition 7-4 for price and output and in Condition 7-5 for abatement. This factor is close to unity as long as a competitive economy with many firms is studied, because the market share of each firm will then be small. The second and main difference is in the supply price of Condition 7-4, which has the refund term, $T(E/Q)(1 - \sigma_i)$. This term shows the output effect: it results in a lower price and thus larger output than with the tax instrument. However, the price will still typically be higher—and output lower—than in a situation without any instrument at all.

revenues of which are returned to the aggregate of taxed firms. Thus, polluters pay a charge on pollution, and the revenues are returned to the same group of polluters, not in proportion to payments made (which would be nonsensical because it would make all net payments zero) but in proportion to another measure, such as output. Thus, the individual polluter pays a tax on its emissions (Te_i) and receives back a share of the total fees collected $[(q_i/Q)TE]$ based on output (q_i), where Q and E are aggregate output and emissions, respectively (see Box 7-3). The aggregate of all firms pays for abatement costs, but it does not pay anything more to society. Instead, the net effect of the payment and refund is that the firms with above-average emissions make net payments to the cleaner-than-average firms.

The incentives for abatement with an REP are essentially the same as with a Pigovian tax, as long as the number of firms is large enough that the market shares are small and the REP instrument thus provides adequate incentives for cleaner technology (see Box 7-3). However, the effects of REPs on output price are different from those of taxes (and auctioned or grandfathered emissions permits). The average cost will be lower with REPs than with taxes (because

of the refund) but higher than if unregulated (because of abatement costs). This comparison assumes that the fee in the REP would be of the same magnitude as the Pigovian tax. (However, this assumption is not realistic, because a tax will meet much greater political resistance than the REP, which results in lower net costs to firms.) In fact, output prices with REPs are similar to those under perfect physical regulation (see Chapters 4 and 11) because industry, as a whole, pays no net fees to society.²

These features are both the strength and the weakness of the REP instrument. An REP provides regulators with a price-type policy that can be used for firms that have the right or the power to stop environmental taxes or other instruments that are too onerous. As long as there are significant technical abatement possibilities, the instrument will work well. On the other hand, if no abatement technology is available, the REP will not act as an output or product tax because the output substitution effect is missing in the same way as with, for example, design standards. If the market structure is endogenous, the tax-refunding scheme might have to be complemented with an entry license to ensure that the refunding scheme is optimal in the long run and does not encourage too much market entry (Fredriksson and Sterner 2005; Cato 2010).

An REP scheme in which refunds are proportional to output is analogous to an output-based allocation of emissions permits (Sterner and Höglund 2000; Fischer 2001; Sterner and Isaksson 2006; Fischer 2010).³ This comparison reflects the function and character of the instrument and shows that policymakers are free to choose a price- or a quantity-type instrument irrespective of who (the polluter or society) has the property rights to the environment; even if polluters have the political power to stop pure taxes, regulators do not automatically have to choose a quantity-type instrument such as grandfathered permits (see Chapter 12 as well as examples in Chapters 3 and 4).

The main advantages of REPs—and indeed of tax-subsidy schemes—over taxes relates to the distribution of costs and thus the political economy of the instrument (see Aidt 2010). Because all firms will pay less and some firms even make money in a REP scheme, it will not create the same kind of compact resistance (and lobbying) as taxes. The REP charge also can be set higher, giving stronger abatement incentives than a tax. The fact that the REP is cost neutral also means that it can be used for a subgroup of polluters without greatly affecting their competitiveness vis-à-vis those who are not included. In some cases, this feature might provide valuable degrees of freedom to the decisionmaker.

Supplemental Reading

Aidt 2010
Bohm 1981
Cato 2010
Farrow 1999

Fischer 2010
Pezzey 1992
Sterner and Isaksson 2006
Yohe and MacAvoy 1987

Notes

- 1 Suppose that the deposit for a can is x , the fraction of cans returned is y , and the refund is z . The system would be “revenue neutral” if the refund were larger than the deposit ($z = x/y$). Regulation can be analyzed as a two-stage process in which the regulator applies a charge T on an emission that can be avoided by the customers’ return of the product to the firm. The firm chooses a refund (which need not be the same level as the charge) to entice the customers to hand in the product. The product might have some reuse value (or the offer to the customers might in practice be part of a deal to buy a new model, as is common with vehicles), and the fact that it is returned helps the firm reduce its environmental charges to the state. Firms tend to offer refunds to customers as long as the tax savings plus reuse and other values are larger than the costs involved (Bohm and Russell 1985).
- 2 This concept can be neatly illustrated for an average enterprise with a fixed rate of emissions ($e = xq$), $de/dq = e/q = x = E/Q$: the new refund term $-T(E/Q)(1 - s_i)$ would be equal to the tax term $Te'_q(1 - s_i)$, which it would cancel, leaving $P = c'_q$.
- 3 In this case, assume that the firm needs e_i permits but permits are allocated depending on the firm’s current output, which means $(q_i/Q)E$ permits. Compared with the zero-revenue auctions of permits (Hahn and Noll 1983), the permits are auctioned but the revenues reallocated to the polluters in proportion to a fixed algorithm (e.g., historic pollution share).

Property Rights, Legal Instruments, and Informational Policies

In the previous chapters in Part II, we discussed many classic policy instruments, such as taxes, zoning, and permits. In this chapter, we turn to a more subtle group of policy instruments that goes to the heart of the structure of the economy and society. We start with the definition of property rights, which may be either private, collective, or other. The actual bundle of rights may be defined or restricted in different ways. We dwell particularly on common property resource (CPR) management, because it is important to the management of ecosystem resources in many countries. The chapter then continues with a section on liability and other legal instruments, which can be seen as restrictions on the bundle of rights mentioned. Legal systems and enforcement vary considerably from one country to another, creating diverse conditions for different instruments. We end this chapter by considering environmental (so-called voluntary) agreements and the role of information provision, environmental labeling and corporate social responsibility.

Creation of Property Rights

The foundation of individual negotiation, bargaining, and all contracts within an economy is the creation or definition of property rights (Amacher and Malik 1996). The evolution of property rights is closely linked to the development of types of scarcity (see Chapter 3); with increased scarcity, value increases and pressure builds to create property rights to the benefits of natural resources (Hanna et al. 1996). The definition of property rights also can be used consciously as an instrument.

In many countries, a considerable degree of skepticism and uncertainty surround the concept of creating new property rights. The defining feature of property rights is that they are perceived to be permanent; it is this inalienability that gives owners the confidence and incentive to make long-term and costly productive investments in their properties. However, lack of flexibility

may entail considerable risk: if a government incorrectly assesses the carrying capacity of an aquifer, a fishery, or a forest area, then it may find that it has given away key national assets (such as water supplies) inappropriately.

Government agencies must strike a fine balance between stability and flexibility. Typically, excessive hesitancy in creating and distributing property titles leads to lost opportunities for positive development. Whereas governments may be considered the best protection against valuable assets falling into “the wrong hands,” they have turned out to be bad owners in many cases. They may be such bad owners because they are too distant to provide constructive management or perhaps prone to corruption. The most common problem is a government’s inability to develop resources efficiently.

Another aspect of “bad ownership” is that government often is perceived as having taken rights claimed by local communities (particularly of tribal or indigenous peoples). In many cases where such rights were informal (i.e., unwritten), government has used its power to assert ownership. Large social movements have formed in developing countries to protest state ownership of forest and other resources. One famous example is the Chipko Movement in India. Chipko, which means “tree-huggers” in Hindi, is a pacifist movement whereby villagers have often triumphed against logging companies simply through passive resistance (i.e., hugging trees).

Despite many examples of bad ownership, the government has reason to maintain ultimate responsibility of natural resources, even if some rights are privatized. Bundles of rights (e.g., of exclusion) are not and should not necessarily be identical across social contexts (see Chapter 3). In some poor countries, for instance, complex webs of local culture and other social norms dictate that poor people have the right to graze their animals on richer farmers’ land after the harvest (Chopra 1991, 1998) (see Box 8-1).

In general, the process of modifying rights is fairly slow, and environmental policy instruments (e.g., tradable emissions permits, transferable grazing or fishing rights) evolve relatively quickly. A permit or quota should for instance be seen as a real title to property¹. In this sense, the development

BOX 8-1 Shaping Property Rights: A New and Sensitive Form of Foreign Assistance

Well-established property rights to land and other resources are a decisive asset for many western countries. In some other countries, these rights either have never existed or were destroyed by colonialism. Some aid agencies (such as Sida, the Swedish International Development Cooperation Agency) try to help countries with processes such as land surveying and property delineation, which are essential steps in the development of private enterprise (which requires private property rights).

The lack of clarity concerning rights to property is the most serious obstacle to business in many parts of the world, from Africa to the former Soviet Union. However, the definition of *property rights* is difficult, and in some cases, such projects have been abandoned because powerful groups were acquiring an unfair share of the rights.

or assignment of new kinds of property rights is a definitive policy instrument of natural resources or environmental management.

During the twentieth century, particularly since the end of World War II, international political organizations such as the United Nations played an active role in defining rights (Wiener 1999). The process started with the U.N. Universal Declaration of Human Rights (United Nations 1948). Although the declaration perhaps was not originally intended to have binding force, its provisions reflect customary international law and have gained binding character as customary international law. This document might be considered the foundation for environmental rights because it guarantees everyone “the right to a standard of living adequate for the health and well-being of himself and of his family” (Article 25:1).

The Universal Declaration of Human Rights was followed by many more declarations, treaties, conventions, agreements, and protocols that span the spectrum of environmental and natural resource issues (e.g., desertification, migratory species, biological diversity, wetlands, endangered species, and hazardous wastes and disposal). Some of the more important environment-related agreements that have been adopted in recent years include:

- Rio Declaration on Environment and Development (June 1992),
- U.N. Framework Convention on Climate Change (May 1992), and
- Montreal Protocol on Substances that Deplete the Ozone Layer (adopted Sept. 16, 1987, and subsequently adjusted and amended).

Another important environmental agreement is the 1982 United Nations Convention on the Law of the Sea (mentioned in Chapter 3), through which coastal countries received extended property rights to adjacent portions of the ocean. Some countries found the process unfair, but on the whole, the definition of property rights in this and the other treaties is generally acknowledged as being a necessary prerequisite to whatever other attempts at management the countries have undertaken.

The U.N. Environment Programme publishes various environmental law materials with a view to promoting a wider appreciation of environmental law. Organizations such as the World Bank and OECD also have helped shape international environmental law. Within the OECD, the Environment Committee and the Environment Directorate were established in 1971, and the “polluter pays” principle was adopted by the OECD Council in 1972: “The polluter should bear the expenses of carrying out environmental protection measures decided by public authorities to ensure that the environment is in an acceptable state. In other words, the cost of these measures should be reflected in the cost of goods and services which cause pollution in production and/or consumption. Such measures should not be accompanied by subsidies that would create significant distortions in international trade and investment” (OECD 1975). Worded in this way, the principle rules out subsidies but does

not necessarily mean that the polluter has to pay the full “rent” on the environmental resource used.

Common Property Resource Management

Rights to common property are particularly important for poor people in many developing countries. The development of property rights in general is a gradual process that clarifies the definition, boundaries, and defense of property (see Chapter 3). As more and more resources become scarce, common property rights are replaced by private property rights (this privatization is often called “the enclosure of the commons”).

Some researchers maintain that common property resources (CPRs) may be a superior institution under certain conditions (Balland and Plateau 1996), primarily for common-pool resources (Ostrom 1990; Stevenson 1991; for an analysis of various African cases in which the introduction of formal property titles failed to increase land productivity, see Migot-Adholla et al. 1993). For instance, when the “services” provided by a certain ecosystem are erratic (e.g., rainfall in some areas) or mobile (e.g., the availability of fish or game), then people may find it in their interest to collaborate intensely. When the “services” provided by a certain ecosystem are meager (e.g., pasture in extremely dry areas), the productivity of the land may be too low to cover the basic costs of enforcing private property rights (e.g., by installing fences). However, this kind of relationship is not defined indefinitely; with new technology, productivity and the costs of enforcement can change dramatically.

One basic criterion for determining the need for CPR management is whether the profitability of a private property rights regime would be lower than that of a common property rights regime as a result of either the excessive cost of private property rights on marginal lands (see Chapter 22) or technological factors that make CPRs more productive (see Hanna and Jentoft 1996 and McWhinnie 2009 for some fishery examples).

Many social scientists have questioned the sustainability and optimality of CPR institutions, arguing that they ultimately will break down because of the temptation to free ride. Individuals vary their behavior depending on circumstances. Some adopt a narrow self-interested perspective and will always be free riders, whereas others will behave selfishly only in certain situations, and still others rely on reciprocity and are thus able to overcome the temptation to free ride. A good deal of experimental work has been done on this topic, and results indicate that norms to support sustainable CPR management may evolve easily, particularly if the proportion of permanent free riders is not too large.

CPRs are important in many developing countries, particularly for the welfare of the poorest individuals (Ostrom 1990, 1997, 1999). This logic follows largely from the fact that the poor have little or no other property.

TABLE 8-1 Prisoner's Dilemma

	<i>A plays a1 ("cheat")</i>	<i>A plays a2 ("collaborate")</i>
<i>B plays b1 ("cheat")</i>	(0, 0)	(-1, 11)
<i>B plays b2 ("collaborate")</i>	(11, -1)	(10, 10)

Note: Values are the payoffs received by Player A, then by Player B.

CPRs also are usually necessary for the protection of sensitive and marginal ecosystems (Dasgupta 1993).

The problems of cooperation, free riding and free access, and CPRs can be illustrated by the "Prisoner's Dilemma" game (see Table 8-1). Known among political scientists as a "Social Dilemma," this game is commonly used to illustrate the problems of free riding in CPR management. It offers an explanation to the overgrazing of resources where ownership control is insufficient. The two players choose one of two strategies: "cooperate" or "cheat" in the management of some form of public good or common property. For both public goods and CPRs, players' selfish actions (free riding) create negative externalities. In this context, collaboration is beneficial. (However, collaboration is not usually a good thing, as illustrated, for instance, by collusion on a market.)

Both parties have a common interest in protecting the resource as long as the other one does; each party receives 10 points if they both collaborate. This "payoff" can be thought of as the reward for modest harvesting with respect to the carrying capacity of the ecosystem. Individually, a free rider can gain 11 points as long as the other player continues to collaborate—but the collaborator will get a catastrophic payoff of -1. Therefore, the collaborator is not likely to collaborate further, bringing both players to cheat and receive 0. The point of the game can best be seen by recognizing the uncertainty for each player about the other player's actions. Player B's actions are unknown to Player A. Whatever Player B does, Player A actually appears to do better by cheating. If Player B cheats, then Player A's payoff increases from -1 to 0 by cheating; if Player B collaborates, then Player A's payoff improves from 10 to 11 by cheating. This effect can be analyzed as a reciprocal externality: the actions of one player affect the payoff of the other.

If Prisoner's Dilemma is analyzed as a static game, then it appears to have no desirable solution. This analysis is often done by looking for Pareto and Nash equilibria². There is only one Nash equilibrium (top left, where both cheat), and Pareto equilibria are the other three outcomes. In this simple static formulation, the game has no solution, which is both a Nash and a Pareto outcome—which is a technical way of saying that the spontaneous outcome of this game does not maximize overall welfare.

Several ways out of the prisoner's dilemma have been discussed and experimentally tested. One key factor is the distinction between playing the game once and playing it repeatedly. Another important factor in overcoming the dilemma is the buildup of institutions for control and punishment and

TABLE 8-2 Prisoner's Dilemma: Solution through State Intervention

	<i>A plays a1 ("cheat")</i>	<i>A plays a2 ("collaborate")</i>
<i>B plays b1 ("cheat")</i>	(-2, -2)	(-1, 9)
<i>B plays b2 ("collaborate")</i>	(9, -1)	(10, 10)

Note: The state puts a "tax" penalty of 2 points on cheating. Values are the payoffs received by Player A, then by Player B.

peer enforcement mechanisms (Ostrom 2006 and Van Soest and Vyrastekova 2006). Table 8-2 illustrates the "perfect" government control policy: By putting a fine of 2 points on cheating, the incentive for cheating is removed and the payoffs are changed so that the game only has one "natural" solution. (The technical term is a "core" solution, which is both optimal—from a purely selfish, individual, or "market" viewpoint—in the Nash sense and collectively beneficial in the Pareto sense.)

The trouble with assuming a perfect state control policy is that state control entails monitoring, information collection, and enforcement that are neither costless nor perfect. An unfortunate but likely outcome of state control is that the state controller starts fining the wrong people at the wrong time, creating havoc with the irrigation system, fishery, or other CPR under analysis. Local people then will be worse off than before; they not only will have to bear the costs of state control but also will have to contend with the consequences of its failures, which may lead to the deterioration of local resources management as well as quality.

Cooperative management of CPRs with peer enforcement may be a better alternative since the cost of enforcement is often lower than for the state for several reasons. First, locally selected monitors have better knowledge of local social and ecological conditions and are more motivated than state monitors. They also may have lower salaries and often can combine monitoring with everyday resource management tasks. In addition, control by local peers commonly is perceived as more legitimate than control exercised by state authorities; it also accounts for the heterogeneity of communities along gender and ethnic lines, which might affect the practice of local-level conservation efforts. Indeed, experimental approaches to understanding intercultural conflict over resources in economic decision-making have found cross-cultural behavioral differences across countries and across ethnicities within a country (see, e.g., Ferraro and Cummings 2006).

The setting of CPR management is much better modeled as a repeated game. The dynamic, repeated game turns out to be quite different from the static, single game. In a repeated game, it is profitable for the players to "invest" in such "assets" as reputation, good relationships with neighbors, and the building of collective institutions. Seven conditions are considered essential for stable CPR management (Ostrom 1990):

- 1 Boundaries are clear, and outsiders can be excluded.
- 2 Rules of provision and appropriation are adapted to site-specific conditions.

- 3 Decisionmaking is participatory (democratic).
- 4 Locally designated agents monitor resources.
- 5 A local court or other arena is available to resolve conflicts.
- 6 Graduated sanctions are used to punish infringements.
- 7 Outside government respects the CPR institutions.

Condition 1, clear boundaries and exclusion of outsiders, is a general prerequisite for any kind of property. Condition 2 concerns adaptation of the rules of provision and appropriation to local ecological conditions; rules (e.g., concerning rights to harvest) that are appropriate for one setting may be inappropriate in another. Conditions 3–7 concern the internal “sociology” of decisionmaking. Rules and processes must be democratic, legitimate, efficient, and effective.

Experimental studies on local management of CPR have analyzed the role of institutions improving efficiency of resource use. In an interesting study, Casari and Plott (2003) explored whether an institution used highly effectively in the Italian Alps for centuries would generate positive results in a laboratory setting. In the Alpine system, the population of the village developed among themselves a contract, subject to approval of the regional government, called “Carte di Regola”—a system for monitoring and sanctioning those who are discovered violating or exceeding patterns of use that the villagers agreed upon in the contract. The contract specified in advance the amount of the fine and the conditions under which the sanction could be inflicted. Besides, villagers reporting violations—at expenses of a certain level of monitoring effort—would obtain a share of the penalty. Casari and Plott’s experiment replicating these rules greatly improved the efficiency of resource use as contrasted with the baseline experiment without sanctioning options. Thus, this study highlights the reaction of subjects to both the cost of sanctioning and to the fee-to-fine relationships.

In a recent study, Hasson et al. 2010 analyze the role of trust in enhancing cooperation in climate change mitigation and adaptation. At local levels (as well as multilateral levels), trust can be fostered through communication, signaling, and commitment devices in negotiations.

Local CPR management is a policy instrument that operates primarily at a decentralized level, but central (or local) government also plays an important role. If the government is prepared to accept the autonomy of CPRs, it can benefit from their good management. Central authorities can aid CPR management by providing the necessary legitimacy and by not interfering too much, as suggested by Condition 7, respect for CPR institutions. In situations without CPRs or where the underlying culture has broken down, the central government might try to revive or recreate communal institutions, but rebuilding generally is much more difficult than sustaining institutions that have already evolved.

Because an essential element of CPRs is the gradual evolution of democratically established principles (Conditions 2–4) that have coevolved to fit a cultural and an ecological context, no “quick fix” can restore institutions that

have eroded. This is not to say that such situations are hopeless; arrangements can be made, perhaps partly on the basis of historical precedent but in a way that suits modern society, technology, and changing ecological conditions. In Turkish Alanja, for example, institutions had broken down, but fisheries were rebuilt over a period of more than 10 years to become successful again (Ostrom 1990).

Liability and Other Legal Instruments

The set of rules that ensures the enforcement of other rules is essential to mechanism design. Some instruments are specifically intended to ensure compliance. Such instruments—criminal penalties, fines, liability, and performance bonds—are commonly called “legal” instruments even though other instruments are also based on laws and definitely are not “illegal.” The distinction between legal and other policy instruments is often related to the degree of responsibility required of individuals who cause injury or economic damage to others (e.g., through their businesses).

In Roman law, a distinction was made between degrees of causation and intent, which range from *causa*, which means “cause” (with unawareness or possibly negligence), to *culpa*, which means “causality and fault or responsibility,” to *dolus*, which is “causality with intent.” When two cars are involved in a traffic accident, for example, both drivers are inherently part of the cause, regardless of who is at fault; however, the consequences (e.g., insurance payments and legal action) depend crucially on the assignment and degree of fault or intent. The determination of fault and intention could affect whether a driver faces criminal charges of reckless driving, manslaughter, or even murder.

The character of penalties or sanctions applied to environmental problems depends on the degree of causality. For new environmental problems, when the polluter could not have known about a hazard, the natural response is to provide information about the threats and possible technical solutions. When the causal relations become more established, the main policy concern focuses on the exact level of abatement or the timing, location, and character of emissions. Next, administrative rules and regulations typically become the main instrument, and economists argue for the use of market-based instruments as their complement or substitute. Regulations (and economic instruments) must be backed up with some ultimate threat in case of noncompliance, and these threats are usually financial (e.g., fines). When a firm has knowingly broken environmental or health laws to make a profit, fines are commonly imposed (partly to remove the competitive advantage that the company gained through noncompliance). When the offense is so serious that human health is impaired or people die, administrative and financial sanctions are generally not considered sufficient; in such instances, the responsible managers or employees may be punished under the criminal law system.

Because risks are complex, policies are often used in combination. Good illustrations of combined policy are found in the field of medicine, in which regulations and rules assign liability and require that doctors and drug companies be correctly insured against malpractice. The appropriate progression and mix of civil (economic) and criminal law depends on the character of the society, the problem at hand, and many other factors. In practice, fines and charges may not be so different from a firm's point of view; corporations may "calculate" losses from "expected fines" (factoring in the extent of monitoring, the probability of being apprehended for breaking the rules, and the probability of losing suits).

Whether companies should be strictly or only partially liable for damages has important implications for their behavior. Negligence or fault-based liability implies that the person injured has a right to compensation only if the party causing the injury has been negligent and has taken less than due precautions. Strict liability means a right to compensation, irrespective of precautions. Both principles should lead to the same level of "care," but the business that causes damage often knows that many people who suffer injury never litigate. Suing an employer or a doctor—not to mention government agencies or large firms—is risky and expensive, and many individuals do not have the necessary resources to hire a lawyer and proceed with a lawsuit. Strict liability gives more rights to injured individuals in particular (and perhaps to the weaker parties in society in general). Strict liability could be viewed as the ultimate policy instrument because it should lead to the internalization of all environmental damages and risks. However, its main drawbacks are that it increases the number of cases of litigation in the courts and perhaps generally hampers all economic activities that entail any risk.

In several countries, legal policy appears to be moving toward strict liability. In the United States, many companies (from the chemical industry to general industry to manufacturers of guns and cigarettes) have been sued for negligence as a form of product liability. (In Europe, firms probably would not be sued, but their pollution would be regulated or taxed by the government.) One perverse element of the U.S. suits is that the companies (and CEOs) find it in their interest to prove that they did not know that the products were dangerous. This approach reduces interest in detailed environmental audits, because audits remove the convenient excuse of ignorance. This problem is more prominent under negligence than under strict liability.

Environmental liability typically involves multiple layers of principal-agent relationships; it starts with the relationship between the environmental protection agency and a company and then continues with the relationship between different levels of management within the company's hierarchy. Many serious environmental outcomes not only are the result of poor management practices but also have a stochastic element that introduces the problem of moral hazard³. Also, in some cases, considerable distance in time or space lies between the "cost-saving" in the plant today and the ensuing (risk of) injury

tomorrow, which may reduce management's attention to environmental cost issues. Companies commonly are analyzed as if they had eternal life and "goals" of maximizing expected profit, but in fact they are run by managers who have short-run goals, and this discrepancy may also introduce a bias against environmental caution.

To complicate matters further, the worst-case scenario is not the loss of value equal to maximum environmental damage but bankruptcy of the firm (which typically is worth less than the value of the environmental damage). This is the implication of limited company liability, an institutional innovation making large risks easier to handle that is often seen as one of the main factors behind economic growth in market economies. However, limited company liability can be abused by entrepreneurs who repeatedly and systematically use bankruptcy to get rid of debts. In the case of large hazards, the situation could have more serious repercussions. The owners of plants that represent particularly large hazards can divide them into separate subsidiaries—legally distinct companies. Such separation allows the parent company to reap the profits but avoid the risks because, in the event of an accident, only one subsidiary goes bankrupt, and damages to the parent company are thus minimized. Swedish nuclear power reactors are one example of this approach. Each reactor can be a separate "company" even if it is 100% owned by larger electricity companies.

Environmental problems commonly involve a fairly long chain of causality that precedes injury. For instance, the waste, effluents, or emissions of several firms may be mixed; chemicals may react in the atmosphere; there may be synergies so that the joint effect of two chemicals may be much worse than the sum of their separate effects; and toxins may bioaccumulate in the food chain. Damage to an ecosystem or to human health may not become apparent until years after the hazardous emissions, and the links to a certain productive activity may be tenuous and difficult to prove. Several companies could have been involved, and some may no longer exist. Emissions may have been released legally, and their hazards might have been unknown at the time of the release. In such cases, strict liability may not be a sufficiently forceful instrument. Liability must also be retroactive, and to be really effective, all the parties involved must be liable.

Literature abounds on these issues. In a comparison of four programs under imperfect information (effluent taxes, physical controls, and two types of liability rules—negligence and strict liability), the strict liability rule was found to be superior to quantity and price control (White and Wittman 1982). Negligence has a potential advantage over strict liability in that fewer cases are litigated, but it also has the disadvantage of higher expected deadweight costs. However, the underlying assumption of the strict liability rule is that the firms or polluters have sufficient assets to cover the costs or losses incurred by liability lawsuits. Similarly, individuals can alter risks privately in many ways, and therefore, it has been suggested that strict liability be retained only for extremely hazardous chemicals (because of the expected severe damages from

human injuries, and because victims are presumed unable to take precautions) (Shogren and Crocker 1999). For less dangerous chemicals, negligence liability might be sufficient. A modified strict liability standard has been suggested for chemicals to reduce the moral hazard problem of the producers. The modification would consist of victim precaution requirements, including knowledge of contamination.

Citizens can sue firms for damage, but doing so may be risky. Firms can fight back because they believe that they need protection against unwarranted and harassing suits by overly zealous environmentalists (and others). Firms may, for instance, respond with a Strategic Lawsuit against Public Participation (SLAPP) to stifle environmental activism. Corporate SLAPPs, which amount to more than US\$7 million a year, are intended to intimidate. Fearing the loss of the constitutional right to petition, several states have or are considering explicitly restricting SLAPP regulations. However, some analysts think that restricting SLAPP regulations will not increase the efficiency of the dispute resolution (Hurley and Shogren 1997).

Whereas the typical CEO would not desire bankruptcy, it is not clear whether threat of it is serious enough to make a CEO internalize the appropriate social cost of decisions made on behalf of the firm. Therefore, policy-makers are faced with several additional policy options, including criminal responsibility for employees, mandatory insurance against environmental liabilities, and liability of the parent company and even lenders. However, insurance is a two-edged sword, because it introduces the risk of moral hazard. Insurance itself may guarantee funds in case of damages but does not necessarily provide incentives to guard against damage.

The use of liabilities also changes the game for the company, bringing in more players and presumably raising the costs of capital. New players are faced with the same information asymmetries as already mentioned for the regulator, and the result is not necessarily a simple and clear improvement. Another possible instrument when risks are considerable is to demand a liability bond, which is a deposit of money to be used for repairs in the event of an accident. This instrument is effective in providing protection but can be expensive for a firm. A firm will enjoy the interest earned on the bond, but depending on the characteristics of the capital market, it may have difficulty raising sufficient capital, and the policy would then be a strong discouragement to investments. Liability bonds have not been used extensively.

For firms whose operations involve a risk of such large environmental accidents that they generally do not have sufficient assets to cover the costs, the main moral hazard problem is that such accidents depend on the firms' efforts to prevent them, which is not observable to the government (Strand 1994). Many governments are considering introducing laws to recover cleaning costs caused by pollution damages, which may cause many banks as well as regulators to suffer adverse selection and moral hazard problems. In particular, banks that finance the firms that cause environmental damages may be

considered liable for the cleanup costs. Hence, these banks play a role as additional insurers (Boyer and Laffont 1997). The conventional wisdom about the lender liability is that penalizing lenders for environmental damages caused by their borrowers will drive up interest rates, thus increasing the cost of capital (Segerson 1993); however, this assumption has been challenged (Heyes 1996; see also Jin and Kite-Powell 1995).

In the presence of adverse selection, the firm's bank does not observe the level of profits but does observe the firm's level of effort to prevent an accident, whereas both profits and effort are unknown to the insurer. Full liability of the bank allows for full internalization of the negative externality but leads to insufficient lending. When the cost of alleviating the asymmetric information is too high, partial responsibility might be better than full responsibility, because it can balance the need to internalize the externality and the reluctance of the banks to lend. Moral hazard exists on two levels: one between the government and the firms, and another between the firms and their workers; it follows that adverse selection also exists on two levels (Strand 1996). More important, the liability rules of firms also apply to their own workers, especially for long-term environmental risk or hazard.

To end on a practical note, government should perhaps consider selecting contractors and licensees in risky industries on the basis of firms that have sufficiently large assets of their own, because this approach might provide the best guarantee of payment to victims in the event of damage as well as the strongest incentive to avoid such damages (e.g., Lewis 2001).

Environmental Agreements

In recent years, "voluntary agreement" or "voluntary approach" (VA) has become a catchphrase, and VAs are promoted as a new instrument of policymaking (see Chapter 21). According to the U.S. Environmental Protection Agency (U.S. EPA 1998), more than 13,000 U.S. firms, nongovernmental organizations (NGOs), and local agencies were estimated to be involved in voluntary U.S. EPA initiatives in 2000. In developing countries, particularly those in Latin America, environmental authorities have also embraced VAs and are rapidly putting new programs in place. For example, in Colombia over 50 VAs were signed between 1995 and 2003, and in Mexico ten such agreements were signed during the 1990s (Blackman et al. 2006a,b; Blackman et al. 2010).

The word voluntary implies that the polluters are not coerced; on the other hand, if abatement is purely voluntary, then this method can hardly be described as an instrument. Presumably, there is always some voluntary level of abatement; an economist would surmise that firms at least clean up to the point at which marginal abatement costs are equated with private environmental damages, which may include secondary loss effects to reputation as well as relationships with neighbors, workers, and customers.

To call a VA an instrument would presume a level of abatement beyond that achieved voluntarily. The word agreement suggests more than a purely autonomous decision by the polluter. In fact, the term VA appears to be used mainly for a form of negotiated (and verifiable) contract between environmental regulators and polluting firms⁴. A firm agrees to invest, clean up, or undergo changes to reduce negative environmental effects. In exchange, the firm may receive some subsidies or perhaps some other favor, such as positive publicity, a good relationship with the environmental protection agency, and perhaps speedier and less formal treatment of other environmental controls. This agreement is formalized in a model in which the polluter agrees to adopt cleaner technology in exchange for more lenient regulation (Amacher and Malik 1998a). The potential gain from VAs, for both parties, can be greater under taxes than under quantity regulation (Harford 2000).

What distinguishes this kind of policy from an ordinary command-and-control kind of licensing or regulation may not be immediately apparent. The main difference may be a cultural and psychological one. For example, the “covenanting process”—that is, the dialogue itself, rather than the formal agreement—has been touted as the feature that makes VAs successful (Glasbergen 1999). Today, most companies have environmental expertise of their own and are conscious of image and public relations issues. They may prefer a new label on what is essentially the same old negotiation with an environmental protection agency. The proactive, voluntary approach may be a good way not only of building public image but also of preempting effort by the agency. By taking the initiative in some areas, a firm may be able to divert attention from other areas and be able to set a level of environmental standard closer to its preference (Maxwell et al. 2000). By winning the public relations war, a firm may be able to focus on issues and solutions of its own choosing. In this sense, VAs are closely related to labeling schemes and corporate social responsibility (CSR) (which we discuss in the CSR section below).

In general, one might say that VAs build on the provision of information, as illustrated by one of the most successful VA programs in the United States: the U.S. EPA’s 33/50 Program. A voluntary “add-on” to the Toxics Release Inventory, the program achieved voluntary reductions of the release and transfer of 17 priority chemicals—33% by 1992 and 50% by 1994—one year ahead of schedule. Furthermore, the positive, voluntary approach may motivate the staff of a giant organization, where not merely the highest echelons of management must be convinced but rather the whole organization must become more environmentally aware.

VAs give (up to a point) property rights to the firm. Firms have to pay some abatement costs, but VAs may be a good way for the firm to avoid instruments that require payment for unabated pollution, such as charges and auctioned permits. Formally, this policy could be modeled as a design standard (Chapter 4). The incentives for abatement under VAs may be correct, but the output substitution effect is missing. VAs are most promising when

the opportunity for technical abatement is good while imperfections (in the product or technology markets) make the use of conventional instruments, such as taxes, difficult (Carraro and Siniscalco 1997; Carraro and Leveque 1999). In addition, VAs are an alternative to taxes when emissions verification is problematic (Nyborg 2000).

VAs might appear attractive in cases when the environmental protection agency does not have sufficient power to coerce the polluter. For example, in developing countries, regulators generally use them to help remedy rampant noncompliance with mandatory regulation (Blackman and Sisto 2006). In industrialized countries, by contrast, VAs are used to encourage firms to overcomply with mandatory regulations or to cut emissions of pollutants for which mandatory regulations do not exist. Yet VAs probably lead to the most abatement when the state does have sufficient power—because then, firms are strongly motivated (for a theoretical model, see Ingram 1999 or Segerson and Miceli 1999); VAs might also have a significant effect where regulatory capacity is weak. We can see why this works. In developing countries VAs usually provide a “grace period” during which no noncompliance penalties are applied to the industry but after which more stringent penalties may be applied. VAs thus change the industry’s dynamic investment pattern by reducing short-term investment while increasing long-term investment. Blackman et al. (2006) find that when the probability of effective mandatory regulation is low and the VA allows for a significant increase in penalties for noncompliance, the latter effect outweighs the former, and the VA can enhance welfare.

The actual performance of VAs is difficult to evaluate, but they very probably cannot be used alone in place of other instruments⁵. Anecdotal evidence is both favorable and critical, typically concerning the behavior of large multinational firms. However, the value of such case studies is limited by “success bias.” One of the main reasons that a company undertakes environmental protection activities is to improve its public image. Thus, a good deal of resources are spent on publishing and disseminating the “good news.”

On the other hand, a natural strategy for individuals affected by pollution is to attack the companies where they are most vulnerable, usually in the public relations arena. One of the first attempts to systematically survey the VAs in the United States concentrated on four major U.S. EPA programs: 33/50, Green Lights, Responsible Care, and Project XL (Mazurek 1999a, 1999b). The Green Lights Program encourages energy-efficient lighting technologies. Several thousand participating institutions are required to upgrade lighting “where it’s profitable and where lighting quality is maintained or improved” (U.S. EPA 2002b, 2002d). Average returns are said to be 25% on investments, so it appears that this program is not really causing “costs” to firms in the economic sense. The Responsible Care Program, started by the Chemical Manufacturers Association in 1988, is the most prominent industry-led initiative in the United States. Companies that choose to participate in Responsible

Care (known as “members”) voluntarily adopt six codes of conduct intended to promote safety and avoid pollution.

The U.S. EPA has introduced a great deal of flexibility in its licensing and control of the Project XL Program in exchange for voluntary commitments to abate over and above the required norm. For Intel, which produces processors for computers, the saying “time is money” is almost an understatement. Every time Intel comes up with a slightly faster processor, having to wait to put it on the market would cost the company a million dollars a day. A new processor may well require a “change of production process” that automatically requires new licensing from the U.S. EPA. For such a company, meeting environmental standards and abatement costs are not a big burden, but delays are costly. Thus, the company is prepared to make significant investments in abatement (which also gives the company a good image) in exchange for fast-track treatment on regulatory approval (see Boyd et al. 1998).

Several of the claimed success stories for VAs appear to be overstated (Mazurek 1999a). Although VAs can motivate and mobilize employees, they do not necessarily lead to expensive abatement programs. Furthermore, VAs raise numerous legal problems vis-à-vis regulatory and anti-trust legislation.

The findings concerning VAs generally also apply to “proactive” or “green” companies. Some green companies may be in sectors that have an inherent interest in environmental awareness (e.g., health products and ecotourism), but some may be green partly because of the preference or foresight of managers or as a preemptive business strategy. The World Business Council for Sustainable Development (WBCSD) is a “coalition of 150 international companies united by a shared commitment to sustainable development ... [that] pursues this goal via the three pillars of economic growth, environmental protection, and social equity” whose members represent more than 30 countries and 20 major industrial sectors. WBCSD has succeeded in attracting the CEOs of some of the largest—and, in some cases, heaviest polluting—industries. One of their main catchwords appears to be that of “Green Growth”: technology and entrepreneurial involvement (together with some harmonized regulation) will pave the way to a sustainable future. As their webpage puts it, “9 billion people living well within the resource limits of the planet by 2050.”

WBCSD has some striking corporate policies or experiments to point to, which seem to validate the claim that the council truly is serious about the environment⁶. Critics still see these examples as a form of “greenwashing,” and a true empirical verification is a difficult task. As mentioned in Chapter 6, it is not always clear how effective voluntary trading is or how it interacts with mandatory regimes. However, the main strength of VAs lies in the creativity generated by employee involvement. By involving employees, the many thousands of small, complex everyday issues that are hard to regulate or tax might be addressed. Even though VAs cannot replace other instruments when it comes to large-scale, costly environmental issues such as global warming or acidification, the motivation and proactivity of some of these companies’ employees might be beneficial.

Provision of Information

All policy instruments require information to function, and public disclosure of information has come to be seen as an instrument in its own right. Public disclosure is a hot topic in the search for effective policy instruments in settings where traditional policies fail for some reason—complex technology and ecology, unequal balance of power between polluters and victims, or ideological differences (see Chapter 21). Depending on the degree of interpretation, aggregation of information, or character of the organization, information disclosure can take many forms, such as labeling, public disclosure, or rating and certification.

Labeling schemes are categorized as Type 1, 2, or 3. Type 1 is a voluntary certification that companies apply for; independent agencies set criteria and evaluate products. Type 2 certification takes place in-house, without fixed criteria or independent outside review. Type 3 certification is the provision of raw data, without interpretation or judgment, but sometimes in the form of life-cycle analysis (LCA). As such, Type 3 labeling provides qualified product information—a detailed description of individual effects—but without outside evaluation. Schemes also may be categorized by the item certified: products, firms, processes, or management procedures.

The number of product labeling schemes has been increasing rapidly. “Organic” certification of food is widespread and probably one of the oldest schemes. Type 1 “green” labeling of products has become popular in northern Europe; the German Blue Angel, started in 1977, was the first national eco-labeling program. In Scandinavia, the Nordic Council of Ministers started the so-called Nordic Swan in 1989, and in Sweden, the Swedish Society for Nature Conservation (an NGO), runs the Environmental Choice independent labeling scheme (see Chapter 24). Other examples of certification programs include the Canadian Environmental Choice, the U.S. Green Seal, the Japanese ECO MARK, and the French NF Environnement (OECD 1997).

Somewhat similar to product labeling are public disclosure, ranking, and rating schemes such as the PROPER initiative in Indonesia (see Chapter 21). However, such schemes rate firms or plants rather than products, and the rating or certifying agent is a ministry rather than an NGO.

Another form of disclosure is environmental certification of firms by ISO 14000 or EMAS standards⁷. This kind of policy is oriented toward management, because the administrative routines and administrative structures of firms (or other organizations) are certified, not the environmental standards or performance per se. ISO 14000 requires implementation of an environmental management system and specifies requirements for establishing an environmental policy, determining impacts of products or services, and planning and meeting environmental objectives through measurable targets. Companies are required to organize a credible procedure for environmental management and aim to continually improve in environmental management. In return, companies are certified, which (at least in some markets)

adds value to the firm by boosting its credibility. Certification is not only a labeling scheme; it is also an instrument within the organization. There is a great deal of inertia in large organizations, and management may have to fight to get its policies implemented throughout an organization. In this respect, ISO 14000 is akin to quality control (ISO 9000), and the two standards seem to be becoming closely integrated.

The various forms of information provision are becoming increasingly popular among theorists and in real-world applications (see for example, Amacher et al. 2004, Bowman Cutter and Neidell 2009, and Evans et al. 2009). Information provision has been referred to as the “third wave” of environmental policymaking (after legal regulation and market-based instruments), and its popularity may be explained by the changing costs of providing, processing, and disseminating relevant information (Tietenberg 1998). In this area, there have been some interesting experiences from natural resources management in both industrialized and less developed countries (e.g., organic farming, forest certification programs). In some countries, labeling is being applied to pollution issues and may be particularly promising in situations where the information and other administration costs of more traditional policies are excessive.

In addition, a range of products from public transportation systems to green electricity producers have been certified. In the case of “green electricity,” the homogeneity of the good implies that the price differential provides a good measure of willingness to pay for the “green” attribute, which has no direct, practical importance for the consumer. In Sweden, the extra value of the green label was somewhat above US\$0.001/kWh or some 5% of the electricity price in 2000–2001 (data were actual offers on the market [Kraftbörsen 2001]). The market share grew from 0 in 1996 (at deregulation) to 5% in 1999.

One reason cited for the success of eco-labeling in Sweden is the presence of strong environmental NGOs and thus consumer demand for green products (OECD 1997). At least two strong rival labeling programs exist in Sweden. The European Commission originally hoped to include all European labeling schemes within the EU scheme, but evidence now indicates that competition between rival schemes is good rather than bad, and thus the drive for harmonization has slowed (Karl and Orwat 1999). If this assumption is true, then part of the success in Sweden might be due to competition. Certainly, competition encourages debate on certification criteria. Because the technical and ecological issues are complex, several sources of independent valuation might provide the optimal information input for the consumer.

On the other hand, industry takes a negative stance toward a plethora of standards. Industry also has some reservations concerning Type 1 labeling (ICC 1996):

- The existence of too many labeling schemes may hamper international trade, particularly if there are many individual national labels, and may thus affect competitiveness.

- Labeling schemes should be based on sound science, including the use of LCA, to avoid creating overly simple criteria. However, LCA is so complicated that it is sometimes impracticable.
- The adoption of labeling schemes should be voluntary, and the criteria should be transparent. Business representatives should be involved in the formulation of labeling schemes.

Interestingly, opinions on the role of external auditors are divided on opposite sides of the Atlantic. The International Chamber of Commerce (ICC) as a whole states that the belief in the necessity of third-party eco-labeling is misguided because there is no reason to distrust the companies' own labeling; taken together, the companies' voluntary efforts and national regulations provide sufficient environmental protection. ICC therefore warns manufacturers and governments to be "cautious" when contemplating the use of third-party labeling. The U.S. Council for International Business, a chapter of the ICC, is much more emphatic in its disapproval. In a disclaimer to the ICC document, the U.S. chapter "cannot endorse this paper because it does not adequately reflect their view that multi criteria-based environmental labeling schemes are based on an inherently flawed concept that results in protectionist trade practices, barriers to environmental progress, and deceptive and misleading information to consumers" (ICC 1996).

Others who might be concerned in principle are the exporters of goods from developing countries who might perceive eco-labeling as a nontariff barrier to trade. This concern was investigated empirically by interviewing North American firms that had either the U.S. Green Seal or the Canadian Environmental Choice EcoLogo (Chua and Fredriksson 1998). At the time of the interviews, the green labels did not appear to be important on the U.S. market, and thus, their existence was hardly a barrier to exports from other (developing) countries to the United States.

The demand for information is increasing, and the costs of supply are decreasing. One of the main functions of policymaking in this area seems to be to provide a structure for information flow. In Part III (particularly Chapter 10), we analyze several conditions and criteria for selecting policy instruments in situations that have uncertainties or information asymmetries. Information provision through labeling, disclosure, or other mechanisms is particularly important in these cases. However, the mere provision of information per se is insufficient, whereas the simultaneous use of information as well as incentives that build on that information improves firm performance (Sinclair-Desgagné and Gabel 1997).

In Europe, there has been some political discussion of officially encouraging labeling schemes by requiring public authorities to give preferential treatment to labeled products. Former EU Commissioner for the Environment Margot Wallström has suggested one of the more drastic schemes: differentiated value-added taxes (VATs), with lower rates for labeled products. The feasibility

of this scheme has been investigated by various national governments within the European Union. That such a relatively strong policy would be considered at all in light of the difficulties often encountered in passing more conventional environmental taxes might seem odd. Sometimes the logic of political feasibility is complex; thus, no possibility should be ruled out completely.

A differentiated VAT would be a strong factor in promoting the eco-labeled goods but also would entail several complications. It would completely change the political nature of the labeling schemes by exposing them to much stronger lobbying and would raise trade-related problems by discriminating against imports from countries where the producers may not have had the opportunity to obtain certification. In addition, the environmental benefits achieved by “sanctioned” firms would have to be proportional to the tax advantages received; how to measure these values would have to be determined. The instrument could turn out to be quite costly without being very precise in targeting environmental effects.

Corporate Social Responsibility

Corporate social responsibility (CSR) usually is defined as “actions that appear to further some social good, beyond the interests of the firm and that which is required by law (McWilliams and Siegel 2001). It arises as a response to the demands from multiple stakeholder groups: customers, employees (including top managers), suppliers, community groups, governments, and some institutional shareholders. CSR actions include, for example, going beyond legal requirements in adopting progressive environmental policies such as recycling, abating pollution, developing clean technology, and also human resource management programs, developing procedures on animal testing, supporting local businesses, and embodying products with social attributes or characteristics. However, in practice, the most visible element of a company’s CSR activities is often its charitable giving or sponsorship programs.

Many managers seem to respond to heightened stakeholder interest by devoting additional resources to promote CSR. Other managers take a cooler stance vis-à-vis stakeholder relevance, eschewing demands for CSR out of a belief that such efforts are inconsistent with profit maximization and the interests of shareholders, whom they perceive to be the most important stakeholder. This divergence in response has stimulated an important debate regarding the relationship between CSR and financial performance—specifically, whether socially responsible firms outperform other companies that do not meet the same social criteria—and about the efficiency of corporate philanthropy: whether firms should simply charge lower prices and allow consumers to make their own charitable decisions.

Theorists will tend to point out that “unnecessary” expenditures must reduce profitability, but on the other hand, the costs involved are quite small,

so dynamic and incentive effects due to reputation and brand value could reverse the effect. Empirical evidence has been very mixed. When it comes to environmental performance, many results seem to indicate that “it pays to be green” and that this relationship strengthens with industry growth (Russo and Fouts 1997; Araña and León 2009). In fact, a reputation for leadership in environmental affairs seems to increase sales among customers who are sensitive to such issues. Consider, for example, the task faced by a consumer in deciding whether to buy “hormone- and antibiotic-free” beef from, say, Whole Foods Market. Given that it is difficult to verify the absence of bovine growth hormones and antibiotics—even after consuming the product—it is essential that Whole Foods establish its trustworthiness in the eyes of its customers. Consumers can overcome the uncertainty related to the unobservable environmental features of many products by tying purchasing decisions to firms’ social responsibility records. Publication of popular handbooks such as *The Green Consumer Supermarket Shopping Guide*, the implementation of certification programs such as Green Cross and the U.S. Green Seal, and public disclosure programs such as PROPER and TRI have all facilitated environmentally informed buying behavior. Consumers now have available the information they need to make choices among products and firms. The development of a proactive environmental strategy within the firm can become in itself a source of market differentiation and profits. For example, Procter & Gamble generally credits part of its profits to its pro-environment corporate behavior (Russo and Fouts 1997). However, an environmental reputation must be built on top of an overall reputation for quality. Once gained, a reputation for pro-environment behavior is itself a valuable asset.

As the environmental strategy takes hold in a firm, we would expect it to redesign its production processes. This typically involves motivating the whole staff. Among the positive effects are that a motivated staff may find unexpected cost savings or improvements related to production routines but also to building and auxiliary systems such as heating and lighting and so on. The motivation and energy unleashed can be a great bonus for the firm, or it can be a cost and possibly a distraction. Such a redesign also tends to involve the acquisition and installation of new technologies, which can also enhance economic performance. Physical resources can be a source of competitive advantage if they “outperform” equivalent assets of competitors. If purchased from a third party, a physical asset itself cannot produce premium profits, as that technology is presumably available to competitors. However, if new physical assets allow a firm to capitalize on and enhance its internal methods for waste reduction and operational and fuel efficiency, they can become a source of competitive advantage. Indeed, when internal routines and know-how accumulate, a firm’s knowledge of pollution prevention deepens and the risk of spills and other environmental accidents decrease (Dean and Brown 1995).

Other motivations behind environmental CSR are legitimation and social responsibility (Bansal and Roth, 2000). A motive of legitimation refers to the

desire of a firm to improve the appropriateness of its actions within an established set of regulations and norms in order to obtain long-term sustainability and avoid fines, penalties, and risks in the future. Thereby, this motivation does not arise from proactive effort but from a reaction to external expected constraints. Since the goal is to minimize risks, firms seek to ensure that their environmental performances meet norms and do not deviate from activities of peers. Therefore, firms adopt a more passive and imitative stance, not only to minimize the risk of moving first into novel production processes and strategies, but also to enhance their legitimacy by imitating successful competitors.

Social responsibility stems from the concern that a firm has for its social obligations and values. A salient feature of this motivation is the concern for the ethical aspects rather than the pragmatic ones. In this case, individual leadership is the main driver behind CSR, with high employee morale the main short-term benefit. Firms motivated by ecological responsibility often choose independent and innovative courses of action, rather than mimicking other firms whose motive is legitimization or competitiveness.

CSR and information about company behavior can act on profitability through a number of separate channels: the markets for products, labor, finance, and through the local neighborhood interactions that we could perhaps call the land market (in the broadest possible sense). It seems that all these channels may have some effect: A good reputation will help promote the products, attract good workers and finance, and it could help ease up relations with permitting agencies and the like. However some channels may be more promising than others. The product market often implies the most direct effect, and potentially gains—or losses—can be big since sales are directly affected. This is particularly true in some areas where customers are sensitive and brand value important—for example, baby food, or food in general. In some countries detergents and the like are labeled (see Chapter 24). On the product market we find that not only are green companies rewarded but their less green competitors can be seriously hurt.

On the labor and finance markets, greener companies may also see favorable effects, but it is more difficult to envisage the nongreen companies being seriously affected. This is partly related to the size of these markets and partly to the way they work. Consider the finance market: Suppose half the investors pulled out of some really nasty companies, temporarily lowering their share prices. The effect would most likely not be long-lived since the other half of the capital market would be more than enough to supply the necessary short-fall. In fact for almost any company, it would suffice with just 1% of the financial markets—and the trouble is that this 1% that does not care about CSR would actually face perverse incentives: They would make more money by acquiring nongreen stocks at a discount!

To sum up, CSR can arise from the demands of many groups simultaneously. The good news is that societies at large are increasingly pressing changes upon firms, and firms are responding to that pressure by improving their

environmental management. The increasing availability of environmental information should induce dynamic changes in social tastes and consumer preferences. As these changes occur, CSR will become more relevant. Mechanisms such as product labeling and neighborhood reputation effects are likely to be the most prominent.

Supplemental Reading

Amacher et. al 2004
Arora and Cason 1994
Bizer and Jülich 1999
Blackman 2010

Mason 2011
We the Peoples 1995
Symposium: CSR and the Environment
2008

Notes

- 1 Sometimes new technology can be vital in establishing property rights. In the Amazon Basin, for example, Brazilian authorities use geographic information system (GIS) maps to monitor annual development. This approach has been effective in allowing the enforcement of land-use regulations (i.e., finding forest clearers so they can be brought to court). Similar technology is needed to make property rights secure (CPTEC 2002; Chomitz 2000).
- 2 The Nash equilibrium is one in which neither player can improve payoff by unilateral action, that is, without the possible reaction of the other player. A Pareto equilibrium is one in which no player could get a higher payoff without a simultaneous reduction in the payoff of some other player.
- 3 In 1984, the Union Carbide plant in Bhopal, India, leaked a massive amount of methyl isocyanate, a poisonous chemical gas. An estimated 1,500–4,000 people were killed and many others permanently disabled. The chain of events leading to the catastrophe is complex, and whether the incident depended on minor operating faults and sloppy standards or possibly sabotage from a disgruntled employee is apparently disputed (Fischer 1996). At any rate, the behavior of employees in this context illustrates the moral hazard issue.
- 4 Some authors categorize VAs as purely unilateral, publicly run voluntary systems and negotiated settlements.
- 5 VAs should be used in conjunction with command-and-control or market-based instruments, and with independent monitoring and public diffusion of results, they can be successful. But using VAs as a substitute for another policy in a deregulated environment creates considerable risks for insufficient abatement. Taxes are a more effective threat than regulation when it comes to coercing companies into VAs (Bjoerner and Jensen 2000). With taxes, firms must not only abate but also pay for unabated pollution.
- 6 Two examples are of interest in this book. Deutsche Bank, one of the world's largest financial institutions and a member of the WBCSD, supports the U.N. Global Compact, the U.N. Environment Programme Financial Services Initiative,

and the environmental management system ISO 14001. One example of Deutsche Bank's commitment to promoting sustainable development is the Deutsche Bank Microcredit Development Fund, which is said to be modeled on the Grameen Bank (see Chapter 29). Another WBCSD member has affirmed support for the Kyoto Protocol: Shell unilaterally fixed a target reduction for itself of 10% by 2002 (compared with 1990 levels) and is testing internal carbon trading, with prices fixed at US\$5/metric ton of carbon until 2010 and US\$20/metric ton thereafter (Shell 2001). However, the emissions reductions discussed by Shell (and other oil companies, such as BP, which has a similar system) are process emissions—not the ultimate total emissions related to the burning of their fuels, which are much larger.

- 7 ISO 14000 is run by the International Organization for Standardization (ISO), and the Eco-Management and Audit Scheme (EMAS) is run by the European Commission. EMAS has higher demands on public disclosure than ISO.

PART III

Selection of Policy Instruments

The choice of optimal abatement level is conventionally given as the intersection of two curves that depict marginal damage and marginal cost of abatement (see Figure 4-1 in Chapter 4). However, the smooth marginal cost curves are an unlikely simplification, for several reasons. First, the abatement technologies are not perfectly known, even to the primary decisionmakers—the agents themselves and the regulators. In fact, regulators must rely on partial and possibly inaccurate information supplied by the agents. It may not be possible to know beforehand because the ultimate abatement technologies—particularly fundamentally clean production methods—have yet to be developed, and sometimes the basic science has yet to be researched. Typically, only a couple of point estimates that represent technology blueprints are available, with some approximate estimate of emissions reductions and costs. If these estimates were used as the basis of a cost of abatement curve, then the curve would be very “fuzzy” and have considerable uncertainty.

Considerable uncertainty is also assumed for the damage costs; they are difficult to estimate because they depend on ambient concentrations rather than emissions. The total pollution load on a given population or biotype determines ecosystem and health effects. Damage is probably also nonlinear and presumably varies strongly with location, time, population density, and interaction with other pollutants. Combining the two kinds of uncertainty

creates an even greater uncertainty about the optimal regulation or tax levels. Some instruments may yield smaller expected costs of error than others, depending on the circumstances. The mere existence of uncertainty concerning the optimal level of abatement or the optimal charge probably will encourage lobbying. The actual policy used will always be the result of negotiations and thus, in comparing instruments, one cannot take for granted that, if used, the instruments would necessarily aim for the same optimum.

The chapters in Part III review several special cases in which theory can suggest how to choose policy instruments. In applied work, the problems of policy selection and design are often seriously complicated by the interaction of many of these factors, which are discussed in isolation. Although the facts in any one case may be highly uncertain, there is a great need for stability and credibility in policymaking. Any policy instrument that lacks credible, long-run commitment will be resisted by both judicial and political means, and if it is not expected to last, then it probably will have little if any effect on investment and management decisions by firms, public agencies, and individuals. This situation is particularly applicable to instruments that entail the transfer of property rights and to tax laws—two of the main mechanisms for potentially effective policymaking. Faced with uncertainty, policymakers often like to experiment with policies and adjust them gradually. This approach is legitimate and understandable but may be difficult, because the adjustment itself may be perceived as a weakness and an invitation to lobbying.

In general, no policy instrument is optimal. Given the complexities of real-life situations, there is always a trade-off between several desirable attributes or criteria such as efficiency, distributional effects, administrative costs, and political feasibility. The kinds of sterile debates in which economists typically argue in favor of taxes and other market-based instruments while lawyers and engineers argue in favor of regulations are clearly not productive. A more constructive way of depicting policy selection is to recognize that several instruments are available and that theories from various disciplines can help determine the choice and design of instruments.

The policy matrix, introduced in Part II, can be used to identify and compare various countries' or sectors' experiences with policy instruments. To further analyze policy instrument selection, it is necessary to relate experience to theory, and the discussion would have to be structured in a policy selection matrix. Such a matrix would supplement the policy matrix in two important dimensions: the criteria for policy selection, and the states of the world or conditions that may be relevant. The criteria include efficiency (in various forms, such as static and dynamic allocative efficiency, but also including efficiency with respect to the use of public funds and their transaction costs), effectiveness, fairness, effects on income distribution, and other aspects related to the distribution of welfare, incentive compatibility, and political feasibility. Among the relevant conditions, some pertain to the technology and ecology at stake, such as the existence of nonlinearities, threshold effects, interactions,

or temporal effects or the spatial distribution of effects. Other relevant conditions concern the economy: market forms (e.g., competition, oligopoly, or monopoly), the structure and availability of information, and the preexisting degree of inequality in distribution. Sociopolitical factors (e.g., the extent of corruption, democracy, or openness) may also be crucial.

Combining all this information into a matrix would make it three-dimensional (instruments, conditions, and criteria). To simplify their presentation, conditions and criteria can be joined; thus, the rows would be combinations of criteria and conditions that are relevant for the choice among the various policy instruments, which could be columns in the matrix. Because of the overwhelming array of policy options, criteria, and relevant conditions, a complete policy selection matrix cannot easily be presented on a single page. However, one attempt at summarizing some of the main points is Table 15-1 (see Chapter 15). In some instances, a subset of selected policies might be compared according to several relevant criteria. Given this impracticality, the policy selection matrix is thus more of a “virtual” matrix or an organizational principle. In that sense, all of Part III is an attempt to describe some of the general traits of the choice a policymaker faces, and the different issues raised all represent rows in such a virtual matrix.

The first criteria to be studied are the classical static and dynamic efficiencies, often economists’ core arguments in support of market-based instruments, at least given heterogeneous or rapidly changing abatement costs. However, heterogeneous damage costs can also be a powerful argument for quantitative regulations. In some instances, exact goal fulfillment can be important, which makes effectiveness a crucial criterion. Chapter 9 covers a range of conditions that may affect static and dynamic efficiency, including technical progress, economic growth, inflation, and ecosystem properties that affect the way rules are designed, giving agents the correct incentives to act in the common interest.

Chapter 10 is concerned with uncertainty (e.g., about the development of abatement technologies or about costs of abatement and damage) or asymmetric information. In Chapter 11, the importance of market structure, the general economic context within which the policy instruments operate, and general equilibrium effects are discussed. Chapter 12 focuses on the distribution of costs associated with the implementation of different policy instruments. This leads the way to the discussion of the political economy of policy instrument choice in Chapter 13, which also includes issues related to enforcement and monitoring. Chapter 14 deals with international aspects of instrument choice (as trade and competitiveness), the analysis of transboundary pollution and climate change. Chapter 15 summarizes the most important conclusions concerning the selection and design of policy instruments.

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Efficiency of Policy Instruments

Current environmental challenges are considerable, and some will be quite costly to address. Hence, it is imperative to select goals that balance costs and benefits reasonably in order to reach those goals at least cost; this concept is the same as the everyday meaning of efficiency. Policy instruments must be used because otherwise market failures impede the market from operating efficiently.

In this chapter, we begin by discussing static efficiency (essentially, profit or utility maximization in a static model) under various conditions—notably, the heterogeneity of abatement costs or environmental damage. We then pass to dynamic efficiency, which is the intertemporal efficiency in the use of a finite and depletable stock, and end with a general discussion of efficiency in the presence of such macroeconomic factors as technological progress, economic growth, and inflation.

Heterogeneous Abatement Costs

Choosing a policy instrument depends on many factors, and one prominent point of contention among policymakers is whether to use market-based policy instruments, such as charges and tradable permits. Market-based mechanisms are generally more efficient than other instruments when pollution is uniformly mixed and marginal abatement costs are heterogeneous—that is, when the pollution and damages are the same but the ease with which firms can reduce emissions differs. When differences in abatement costs (or in resource management efficiency) are large, the companies that hold a comparative advantage (i.e., can abate at a lower cost) should be responsible for the lion's share of reparation, and the market is the best instrument to allocate the appropriate tasks. This idea is obvious to most economists. However, because it is an essential component of environmental and natural resources policy, the topic warrants further discussion here.

Figure 4-1 (see Chapter 4) is a classic diagram in environmental economics, but its simplicity is misleading. It illustrates two curves: marginal cost of abatement and marginal cost of environmental damage. These cost functions are

drawn as curves or sloping lines rather than fixed cost levels (flat, horizontal lines) because the variation in costs with emissions is important. The marginal approach is an important contribution from economics that also is relevant to the understanding of policy design. There is no such thing as “the cost” of abatement or of environmental damage; both values depend on the level of emissions or emissions reduction, respectively. Thus, the (implicit) price of pollution and pollution prevention vary—often considerably—with the intensity of pollution. Typically, the marginal damage from pollution increases with emissions (an upward slope). However, after attaining a certain level of damage, the curve may flatten or fall toward zero. This means that a point has been reached at which additional emissions cause little additional damage (e.g., a river may cease to be “a river” in the traditional sense and become “a sewer”). Similarly, the marginal cost of abatement typically increases with increased abatement, which is the same as decreasing with increased emissions (a downward slope). Indivisibilities, varying returns to scale, and technological development may modify these typical curves in individual cases.

The point at which these two curves intersect is the optimum. With more emissions, the marginal damage is higher than the marginal costs of abatement, and so a cleaner environment is preferred. To the left of the optimum is the opposite situation: the marginal costs of abatement are higher, and thus the cost of continuing to clean up the environment is more than the benefits are worth. Environmentalists might find it difficult to accept, but the resources that would be required to further decrease emissions could be used more efficiently if spent elsewhere in the economy, which includes spending on other environmental issues.

If environmental damages are external (e.g., because ownership rights are not properly defined or enforced), then the market will not arrive at this optimal level on its own, and policymaking may have a role (“may” because such a policy makes sense only if it does not introduce new distortions or costs that are worse than the externality it is intended to correct). Under many overly simple conditions, it would then not make any difference if a charge (T^*) or a quantitative restriction (e^*) on emissions is implemented. However, determining the optimal level of abatement is not so simple if you consider that this abatement has to be allocated among many firms. One of the main reasons why the curve for the marginal cost of abatement slopes upward might not be that each firm finds successive abatement more expensive (although this certainly can be the case) but that different firms have different abatement costs.

The first step in modeling this scenario is to introduce heterogeneity into the abatement equation. To do this in a simple manner, consider two firms that have abatement functions of the form $MC_i = h_i a_i$, where MC is marginal cost, h is heterogeneity, and a is abatement as defined in some appropriate physical measure. For clarity, normalize h_1 to 1 and set $h_2 = h$ as the key heterogeneity variable to indicate how much more expensive abatement is at Firm 2 than at Firm 1, thus assuming that $MC_1 = a_1$ and $MC_2 = ha_2$. The (optimal) aggregate

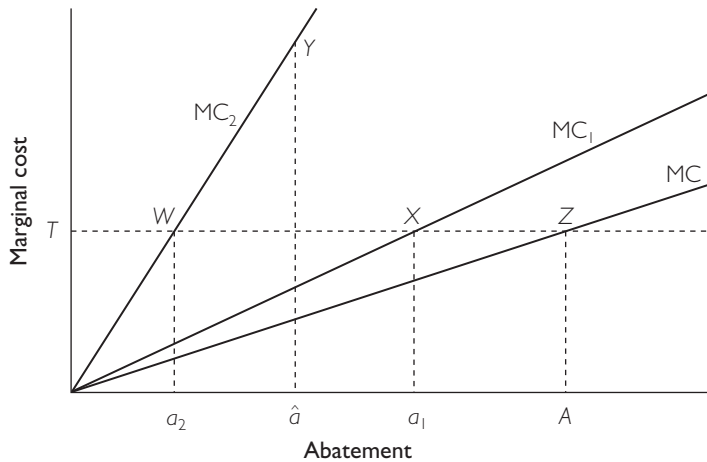


FIGURE 9-1 Static Efficiency with Heterogeneity in Abatement for Firms 1 and 2

Notes: MC_1 and MC_2 = marginal costs for Firms 1 and 2, respectively; MC = aggregate marginal cost; T = tax level; a_1 and a_2 = levels of abatement that equalize marginal costs for Firms 1 and 2, respectively; $\hat{a}$ = equal abatement for both firms; A = aggregate abatement ($a_1 + a_2$, or $2 \infty \hat{a}$).

abatement is derived from these individual abatement curves and labeled as MC in Figure 9-1. Emissions are assumed to be uniformly mixed, which means that the sum of the emissions causes the environmental damage; otherwise, emissions could not simply be traded between sources (the case with heterogeneous damages is discussed below). The intersection of the curves for the marginal costs of pollution damage (MCE) and aggregate abatement costs (MCA) determines the optimal levels of emissions and their shadow cost, T^* (see Figure 4-1 in Chapter 4).

Figure 9-1 shows the allocation of abatement between the two firms. Either abatement could be mandated at a target level of abatement ($\hat{a}$) for each plant (abatement levels are assumed to be equal here, which might be a reasonable approximation of regulation if the firms were of equal size¹), or a market-based policy instrument such as a charge (T) could be used, giving abatement of a_1 and a_2 , respectively. This solution is the most cost-efficient, and the same result can (under ideal assumptions) be achieved by allowing the firms to trade their rights to emissions reductions. Under the trading scenario, Firm 1 (with lower abatement costs) will voluntarily undertake more abatement to create and sell surplus emissions credits or rights to Firm 2, which finds that it is cheaper to buy these on the market than to incur the costs of abatement.

The total abatement costs to the companies are the areas under the MC curves in Figure 9-1. When reductions in abatement are equal, costs are the sum of two triangles with base $O\hat{a}$, of which one ($O\hat{a}Y$, costs to the company for which abatement is difficult) would be large. Under a market-based allocation scenario, the company with lower abatement costs is responsible for most of the

abatement, and thus aggregate costs are significantly lower ($Oa_1X + Oa_2W$). If there are big differences in the abatement cost functions of the two firms, then the equalization of MC of abatement can reduce aggregate costs significantly.

Savings from Market-Based Instruments

Although the exact savings depends on several factors, including the number of firms and the form of the abatement functions, the general principle can be illustrated with a simple example. For the function presented above, in which the marginal cost of abatement increases linearly with emissions reductions, assume that the regulator has chosen a fixed target for emissions reductions. The potential cost savings from market-based instruments (MBIs) compared with an equal reduction are listed in Table 9-1.

Thus, if $h = 1$ and the abatement costs of the firms are identical, then the costs are the same for the two policies and equalization of marginal cost will not yield any savings. With small differences in abatement costs, the savings are minor. Even with 50% differences in the firms' abatement costs ($h = 1.5$), the potential savings from MBIs would only be 4% compared to that with an insensitive regulation that required equal abatement. However, when differences in abatement costs are substantial (e.g., one or two orders of magnitude), the savings can be significant. Although such results depend on a special set of assumptions (i.e., two firms with linear marginal abatement

TABLE 9-1 Cost Heterogeneity and Savings due to Efficiency

<i>Heterogeneity</i>	<i>Cost</i>		<i>Savings by MBI (%)</i>
	<i>Equal reduction</i>	<i>Efficient reduction</i>	
1	2	2	0
1.5	2.5	2.4	4
2	3	2.67	~11
3	4	3	25
4	5	3.2	36
9	10	3.6	64
99	100	3.96	~96

Notes: MBI = market-based instruments. Cost values are indices of cost, which must be multiplied by $\hat{a}^2/2$ to get actual costs. Assume a target reduction in abatement of $2\hat{a}$. An equal abatement reduction plan with $a_1 = a_2 = \hat{a}$ would cost $\hat{a}^2/2$ for Firm 1 and $h\hat{a}^2/2$ for Firm 2, giving a total of $(\hat{a}^2/2)(1 + h)$, where h is heterogeneity. Equalization of MC would imply $MC_1 = MC_2$ and thus $a_1 = ha_2$. Because $a_1 + a_2 = 2\hat{a}$, the emission reductions would be $a_1 = 2h\hat{a}/(1 + h)$ for Firm 1 and $a_2 = 2\hat{a}/(1 + h)$ for Firm 2. The costs for the two firms would be $2h^2\hat{a}^2/(1 + h)^2$ and $2h\hat{a}^2/(1 + h)^2$, respectively. The total cost with equalized marginal costs can be written as $(\hat{a}^2/2)(4h)/(1 + h)$. A comparison of total costs indicates that equal emission reduction is more expensive than the least cost (i.e., equalized MC) by a factor of $(h + 1)^2/4h$, which gives the cost savings for different values of h listed in the table (see Box 15-1 in Chapter 15 for more details).

cost and a fixed abatement target), they illustrate how strongly the savings from the static efficiency of MBIs are related to the underlying heterogeneity of the abatement cost functions.

Because the so-called static efficiency property of MBIs is one of the strongest arguments for their use, it is important to consider when and why abatement costs might differ. Some fundamental reasons are related to the timing of abatement investments. The price of abatement technology may decrease over time or with increasing experience. If the abatement technology is developed by one of the firms in the industry, incentives for sharing technology with competitors may be complicated, depending on the policy instrument used (Höglund 2000). The cost of abatement may depend on capacity use in the abatement equipment industry; the cost may be artificially (and temporarily) high if many firms are making similar investments.

If abatement investments are complementary with other investments or if the abatement technology is embedded in other capital, then abatement investments can be timed to coincide with other investments the company is planning, thus lowering costs of abatement. The cost of complying with an environmental regulation in connection with a new investment or a major reinvestment is often a fraction of the cost of retrofitting a plant at any other time (which probably would require closing the plant and halting production; see Box 9-1). A similar difference is found in the costs of reducing emissions from transport. Reducing emissions at the moment of purchase (or design) of a new car is relatively inexpensive, as is retiring old and polluting cars. However, reducing the level of pollution emitted from the average five-year-old car is an expensive undertaking (see Part IV).

A related issue about timing is that the costs of an abatement investment depend heavily on whether there is slack capacity in management, engineering, and production resources of the firm. A large share of the cost is opportunity cost (i.e., productive work forgone); if management or technological experts in the firm are fully employed in productive activities, then the time spent on compliance with a new regulation may be a considerable opportunity cost.

BOX 9-1 Cost and Timing of Abatement Investments

The relationship between cost and timing is nicely illustrated by vapor recovery systems for gas stations. The cost of equipment for gasoline vapor recovery is almost negligibly low, and the gasoline saved (as condensed vapor) as a consequence of its use provides some long-term cost savings for its owner. The net cost of installation of such a system in a new gas station or in a station already undergoing major renovation is estimated to be quite negligible (around 0.01 cents/liter) (Katz and Sterner 1990). The cost of closing a station for a week and tearing up the asphalt, however, is sizeable—particularly for a small station—and could be a hundred times higher, around 1 cent/liter. Consequently, retrofitting a gas station to comply with a regulation on short notice may be prohibitively expensive.

These costs could have been avoided if the company had had the leisure of choosing the time for compliance. On the other hand, polluters cannot be allowed to choose their own timing either, because they would then have too strong an incentive to postpone action.

Cost heterogeneity may result from plants of various ages in an industry, or from the emission of the same pollutants (greenhouse gases, lead, chlorofluorocarbons [CFCs], and solvents) by several sectors. The inherent differences in scale, type of technology, and other factors increase the chances that abatement costs will differ. For example, it was easier to phase out the use of CFCs as propellant gases in spray cans than in some areas of refrigeration and cleaning. In such cases, MBIs may be more efficient, although a common-sense flexibility in the use of regulations—such as banning CFCs or lead in one area of application long before they are banned in another—can capture a large share of benefits, too. The speed of technological development and thereby cost reductions in different areas also may be affected by the choice of instrument.

Empirical evidence shows that abatement costs are heterogeneous in many cases, thus suggesting that MBIs are warranted. For instance, variations in actual abatement costs in the United States for various classes of air pollutant reportedly vary by a factor of at least 5–10 (Hartman et al. 1994).

Heterogeneous Damage Costs

The counterpart to heterogeneity in abatement costs is heterogeneity in damage costs. In the model of heterogeneous abatement costs, pollutants are assumed to be uniformly mixed. When damage costs are heterogeneous, the damage of a certain pollutant varies depending on the timing, location, or other circumstances of its emissions. For simplicity's sake, in this section, abatement costs are assumed to be homogeneous. Damage costs could be heterogeneous for two reasons: either essential characteristics of the pollutant or emission are different, or decisive variations in the ecosystem characteristics of the environment determine the level of damage caused by a given disturbance.

If pollution is purely local and each pollutant unique, then the argument that one enterprise may be better at abatement than another is no longer relevant. If gold mining leads to mercury emissions that affect one river and the disposal of bagasse from sugarcane production affects another river in a completely different way, then there is no point in discussing the relative ease of abatement or of trading rights between the two cases. Reducing pollution of one type in one locality simply will not have any effect on the welfare of individuals in the other locality.

For health costs, all else equal, the number of people affected will increase the estimated external costs of emissions. This assumption applies to any industrial or transportation pollution whose damage costs are related to population density (discussed in Part IV). This heterogeneity is different from

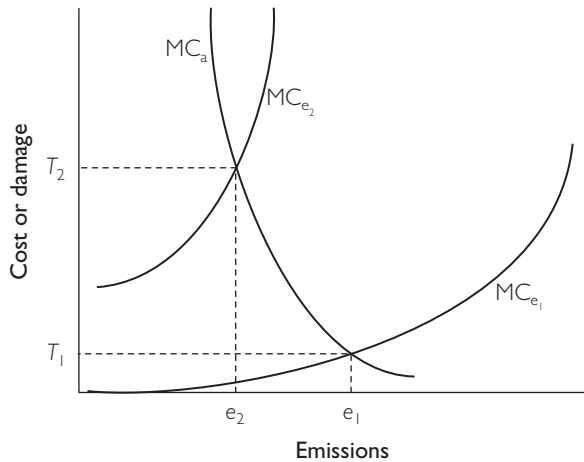


FIGURE 9-2 Efficiency with Heterogeneity of Damage Functions

Notes: T = tax level (for Firms 1 and 2); MC_e = marginal damage due to emissions (of Firms 1 and 2); MC_a = marginal cost of abatement.

heterogeneity in abatement costs because the optimal charge itself varies with the cost of emissions (Figure 9-2). The optimal level varies because there is not one single market for pollution or abatement but several geographically separated ones. A familiar example is the availability and price of parking, which has considerable spatial variability. An early and environmentally important example was estimated gains from spatially differentiated environmental charges for managing the estuary of the Delaware River (Kneese and Bower 1968). Equal taxes may be relevant for global (mixed) pollutants whose only effect is global warming (e.g., carbon dioxide). However, most other pollutants are, to varying extents, both local and regional in effect, and their tax ideally should be differentiated.

Although differentiated taxes may be feasible in some cases, they may be difficult to implement in others. Regulations also would have to be differentiated rather than homogeneous. However, differentiated regulation (or zoning) is often considered to be much more acceptable than differentiated taxes, which bring differences in not only the degree of abatement demanded (hence, in abatement costs) but also tax payments for unabated pollution. These taxes may be much more important than the difference in abatement costs, as illustrated in Figure 9-2 (see also Chapter 12 on the politics of cost distributions).

The concept of “equal taxes” is deeply engrained in culture; it often is protected as a principle by law and, in some countries, in the constitution. Inevitably, urban polluters confronted with a differentiated tax would complain that their rural counterparts are unduly advantaged if the urbanites are more heavily taxed “only” for living in a densely populated region. (The counterargument is that this approach is akin to other market prices: land

is, for example, more expensive in cities without being considered unfair.) Similarly, area pricing fees that charge motorists entering an (urban) zone will inevitably be thought of as an unfair “entry fee” by motorists from rural areas.

Taxing a polluter for certain emissions and charging rates that reflect differences in location, time, and other variables requires sophisticated monitoring. In many cases, actual emissions cannot be monitored or taxed; instead, the tax is levied on an input or output that is assumed to be a close complement or cause of the pollution (e.g., leaded gasoline), which makes it more difficult to differentiate the tax with respect to timing or geography of emissions (see Chapter 5). Such is the case if transport costs of the taxed good are small and no other monitoring of use or emissions is possible. For instance, differentiating the price of gas to reflect external effects inside and outside a city is difficult because consumers are mobile. However, water prices are increasingly being differentiated. As more sophisticated meters become less expensive, decisionmakers are realizing that this approach may be one way of internalizing the costs of pollution and sewage treatment associated with water consumption (see Chapter 23).

When might damages be expected to be heterogeneous? One example has already been mentioned: the health effects of local air pollution. Instruments such as zoning are applicable to many local pollution (emissions, noise) problems that depend on the number of people affected. Zoning can significantly reduce many externalities at an administrative cost much lower than regionally differentiated taxation or ambient permit trading. Zoning (or, more generally, geographical separation) of economic activities that have external effects on each other or on third parties can substitute for or complement other policies to reduce the external effects. For instance, if the emissions from a paper mill damage fisheries, agriculture, or some other industry (and possibly vice versa), then conceptually, three kinds of policy might improve the situation: relocating the firm that causes emissions, relocating the activities that suffer from the emissions, or abatement. In such situations, a partial analysis usually is not sufficient to determine the optimal policy combination; a total analysis of all the possible policies may be necessary.

Damage costs also commonly vary with other factors, such as time of day and weather. The time of day has obvious importance for a valuation of noise and may affect the cost of air pollution, because the number of people affected typically changes throughout the day. The damage curve is often nonlinear, so the marginal cost of an extra unit of pollution is higher in heavily polluted areas (or perhaps lower—pollution over a certain threshold may have little additional effect).

This nonlinearity, together with the resilience of the ecosystem affected, forms the basis of the concept of critical loads of pollution. Whereas some areas may be able to tolerate additional acid rain or runoff (e.g., of pesticides or metals) without incurring significant damage, the damage might be considerable in other areas. For example, the buffering capacity of underlying

rock varies so much that the damage from acid rain in much of Scandinavia is several orders of magnitude worse than in areas such as England, where bedrock contains primarily clay and chalk. Similarly, some weather patterns (e.g., inversion) cause pollutants to accumulate, and thus additional pollution may have a greater impact on those days. Another reason for time differentiation is the daily rhythm of the labor market and urban transportation, which cause congestion and the rapid buildup of pollutants just when the concentration of people affected is at its highest. As a result, the environmental damage cost of driving a car in an urban area typically varies strongly depending on the time of day; therefore, restrictions and congestion fees should be designed to reflect this variance (see Part IV).

Another reason for differentiated policy is the plethora of pollutants. Innumerable chemicals have different properties depending on the ecosystems into which they are emitted. Thus, the number of damage curves to consider is large. Given that knowledge and resources are inherently limited, considerable uncertainty will always surround the determination of appropriate levels of emissions and taxes. Synergies among pollutants are not uncommon, so the total damage from all pollutants is nonlinear. Thus, the estimate of the damage curves will vary with changes in knowledge and changes in pollutant concentrations.

As long as pollution permits are not tradable, differentiating the permits by geographical area is simple in principle. However, to reap any of the gains from specialization or trade mentioned in the previous section, permit trading must be allowed. If pollutants are perfectly mixed, then trading is a natural solution. If each instance of pollution is completely separate, then trading makes no sense. The intermediate case—that is, when there is some mixing of pollutants, or when each emission has both local and regional or global effects—is the most common. The more complex realm of ambient trading schemes, or differentiated taxes, is discussed in Chapters 4 and 5.

Damage curves may be fairly steep and shift frequently, even stochastically, with an exogenous variable such as time of day, weather, or other social or ecological factors. Examples of such damage include atmospheric emissions, for example, from vehicles. The level of damage may depend strongly on population density and weather. Similarly, the effect of a certain level of fishing activity may depend on the current fish stock. The fish stocks vary stochastically—in some cases, chaotically—and thus, a level of fishing that may be acceptable when stocks are high might lead to rapid depletion when stocks are low.

These examples illustrate situations in which the appropriate level of an activity should change often. The chosen instrument must be flexible. The authorities that have the relevant knowledge must also have the power to fine tune the policy instrument frequently. This thinking goes against the traditional kind of license, but many precedents exist for constructing rules that vary according to certain predetermined characteristics: allowing noise only at specific times of the day, differentiating speed limits by route, and tailoring

TABLE 9-2 Policy Selection with Heterogeneity in Both Abatement and Damage Costs

	<i>Homogeneous damage costs</i>	<i>Heterogeneous damage costs</i>
Homogeneous abatement costs	(Either policy)	Individual permits, zoning, and other restrictions
Heterogeneous abatement costs	Charges, taxes, or tradable permits	Differentiated input prices; local trading of permits in hotspots

emissions levels to specific chemicals. In the fishing industry, a common policy solution is to denominate the transferable catch rights not in tons but in percentages of a total allowable catch, which then can be altered by authorities without affecting the relative competitive positions of the participants (see Chapter 25).

Price instruments also can be varied. Because taxes have to pass through a complex legal and political procedure, they are generally thought to be less suitable when fast and frequent changes are required. Fees or charges do not have to go through the same kind of procedure and should be much more flexible; however, in many countries, the right of local authorities to change tariffs and charges as they please is severely restricted.

The treatment of heterogeneity in damage costs is particularly interesting in China, where pollution charges are supposedly equal across the country (for the sake of fairness). In fact, implementation is uneven, so the effective fee turns out to be significantly higher in some parts of the country than in others. The more densely populated and affluent areas tend to pay more than other areas, which may well correspond to a reasonable *de facto* differentiation of the fees to reflect actual damage costs (Wang and Wheeler 1996) (see Chapter 21).

When both damage costs and abatement costs are heterogeneous, policy-makers must base their choice on the relative strength of the two sources of heterogeneity as well as the degree to which each is observable (see Chapter 10). Sometimes both issues can be addressed simultaneously, for example, if input prices can be differentiated locally or if a permit trading system can be supplemented with local restrictions in the areas where pollution is worst. If the damage costs are high in cities, then the regulations would be strict there, and no trading of permits would be allowed between sources in the city and sources outside the city unless a special “exchange rate” were used. If abatement costs vary among the plants in the city, then large savings in abatement costs can still be achieved by trading permits among those plants. If polluting inputs have high transportation costs, then they, too, can be priced higher in the cities (see Table 9-2).

Efficiency in an Intertemporal Sense

Natural resources economics focuses on the management of stocks. For pollutants that accumulate, the stock rather than the flow of pollutants causes damage.

Examples of such pollution range from the buildup of toxic metals or persistent organic chemicals (such as DDT and PCBs) in the food chain to the emission of greenhouse gases (such as carbon dioxide) that cause climate change.

Most pollutants cannot be categorized as “pure stock” or “pure flow”; they have elements of both types of effect. Noise is a good example. Technically, it is a pure-flow pollutant because noise is momentary and does not accumulate. However, medical health effects such as hearing loss and stress are linked to cumulative exposure; therefore, the problems caused by noise pollution may have some stock characteristics. Also, one problem can have several causes (e.g., global climate change is caused by several gases that persist in the atmosphere for different lengths of time). To a large extent, carbon dioxide has stock characteristics: an increase in carbon dioxide leads to some increased assimilation; thus, the long-term increase is not equal to the short-term increase, but decay takes a long time. For water vapor, ozone, or nitrogen oxides (NO_x), the assimilative processes are much more rapid and thus have more of a “flow” character. Pure stock pollutants essentially use up a finite resource (i.e., the assimilative capacity) and can thus be analyzed analogously to the optimal use of natural resources. Regulators might impose or create some kind of scarcity rent to compensate for the fact that no owner or market mechanism does this automatically (see Chapter 22).

A scarcity rent can be captured either by gradually raising charges or by implementing physical constraints that become increasingly binding over time. Both options have the potential to work, but they bring about different consequences. A third option is to create property rights, as has been done in some fisheries (see Chapter 25). Some stock pollutants (such as DDT, which accumulates in the food chain) have been banned completely in some countries because the hazards are large and substitutes exist.

MBIs must be used for pollutants whose costs of abatement vary significantly (e.g., atmospheric carbon). Making a distinction between stock and flow externalities can shed some light on the choice between price-type instruments and regulation for global warming (Pizer 1999). The general starting point of the analysis is that because climate change has a stock character, the benefit curve for emissions reductions is fairly flat, which favors price-type instruments². In addition, the optimal control path for emissions depends on the relative slopes that tend to further favor price controls. However, this result can be modified and counteracted by several factors, including low rates of discount, low rates of stock decay, and positive correlation of cost shocks across time.

In some cases, other factors such as administrative simplicity, scientific criteria, or international agreements have led to the explicit adoption of quantitative goals. The existence of such quantitative targets may in itself be an argument for regulation or tradable permits over taxes—for instance, in application to climate gases where complex international bargaining appears to focus on percentage reduction targets for each country. Such targets suggest an advantage for permit trading over taxes also at the level of national

policymaking. However, the experience of congestion pricing in Singapore indicates that iterative changes in tax may sometimes be chosen, even if a quantitative target is being pursued (see Chapter 16).

To choose policy for ecosystems with more complex characteristics (e.g., those discussed in Chapter 22), not only stock aspects and intertemporal optimization but also risk and uncertainties related to irreversibilities and interacting nonlinear effects must be considered.

Technological Progress

Technological change has become a major focus in environmental policy since it is assumed to yield a double dividend: not only would it stimulate economic growth, it would benefit the environment by reducing emissions. Technological change is also the main force assumed for instance in the Porter hypothesis—the idea that environmental regulations spur innovation and competitiveness.

Technological progress is a very strong force. People who have worked with or even casually observed environmental protection over time have observed many instances in which polluters first claim that they cannot reduce emissions, only to proudly announce a short time later that they have found a way to cut the same emissions drastically³ (see Table 9-3).

Schumpeter (1942) distinguished three stages in the process by which a new, superior technology permeates the marketplace. Invention constitutes the first development of a scientifically or technically new product or process. Inventions develop into an innovation when the new product or process is made available on the market. The invention and innovation stages are carried out primarily in private firms through a process that is broadly characterized as “research and development” (R&D). Finally, a successful innovation is diffused when it becomes gradually available for use in relevant applications

TABLE 9-3 Emissions Reductions for Selected Pollutants in Sweden, 1990–2008

<i>Substance</i>	<i>Reduction (%)</i>
CO	44.5
NO _x	49.0
Mercury (to air)	65.9
Cadmium (to air)	77.9
Lead (to air)	97.4
SO _x	70.9
VOCs	50.9
Arsenic	85.1

Notes: CO = carbon monoxide; NO_x = nitrogen oxides; SO_x = sulfur oxides; VOCs = volatile organic compounds. *Source:* CEIP (2010).

through adoption by firms or individuals. (For an environmental, empirical application of the innovation and technology diffusion created by strong policies on NO_x in Sweden, see Sterner and Turnheim 2009).

The market failures related to the process of technological change interact with the market failures associated to environmental pollution. A firm that invests in R&D or adopts a new technology typically creates benefits to others while incurring all the costs. The positive externality of innovation comes from the public nature of new knowledge and the existence of “knowledge spillovers.” Since it is difficult to exclude others from the benefits of innovation, firms do not have enough incentives to undertake the necessary investments. Although patents and other institutions might protect a firm’s investments toward innovation, such protection is inherently imperfect. On the other hand, since information about the prospects for success is asymmetric (the innovator is in a better position to assess the innovation’s potential), investors might be skeptical about the promised returns and demand a premium for investment, discouraging innovators.

In the case of technological adoption, for a number of reasons the cost or the value of adopting a technology to one user may depend on how many others have adopted the technology. For example, the adopter of a new technology creates a positive externality for others by generating information about the existence, characteristics, and success of the new technology—a process usually referred as “learning-by-using.” Production costs might also fall as manufacturers gain production experience; this supply effect is known as “learning-by-doing” and implies that the cost of adoption decreases as the fraction of firms using the technology increases. Finally, there might be also “network externalities” that imply that a technology becomes more valuable to an individual user as other users adopt a compatible technology.

Thus, the interplay of technology and environmental market failures implies that markets underinvest in new technology. It is unlikely that environmental policy alone creates sufficient incentives for technological change, adding weight to the case for second-best policies or for implementing further policies. Indeed, raising taxes above the Pigovian level could boost emission-savings technologies but at the expense of decreased output (see Hart 2008). On the other hand, there is a long history of public support for research in the United States and other industrialized countries (see Jaffe et al. 2005 for further discussion). Innovation policies include public supply through research done in public institutions, subsidies, or matching funds provided to firms for specific research proposals or joint industry-government research. Technology adoption can be encouraged with tax credits that reduce the effective purchase price of new equipment that meets specific criteria or with information programs to increase awareness about the availability of environmentally friendly technologies (Anderson and Newell 2004; Jaffe et al. 2005). Finally, technological standards can also be used to encourage the diffusion of particular technologies (see Chapter 4). For example, the Corporate Average

Fuel Economy (CAFE) standards have been designed to induce an improvement in vehicle fuel efficiency (see Chapter 17). Although such standards can be beneficial, they can also go beyond economic efficiency if the population of adopters is very heterogeneous.

Policy Instruments for the Adoption of New Technologies

When comparing the incentives for innovation, adoption, and diffusion of different policy instruments it is easier to begin with the incentives for adoption. Firms engaging in R&D of abatement technologies are either part of the polluting industry itself or engage in R&D in order to sell or license their new technology to other firms. In the first case, the innovator's benefit is given by the change in compliance costs and profits in the output market plus the benefits it could obtain by accruing license fees. In the second case, the profits depend on the polluting firms' willingness to pay for the new technology, which will be determined by the profits those firms gain if adopting the new technology. However, in both cases the innovator will not appropriate all the rents from innovation if an imitation technology is available (Fischer et al. 2003).

Let us assume that the abatement cost function $C_0(e_i)$ represents the cost to reduce emission of a firm in a competitive market from the laissez-faire emission level $e_i^{\max}$ to some lower level $e_i < e_i^{\max}$. The abatement cost function satisfies the following properties: $C_0(e_i) > 0$, $-C'_0(e_i) > 0$ and $C''_0(e_i) > 0$ for $e_i < e_i^{\max}$ and $C_0(e_i) = 0$ for $e_i \geq e_i^{\max}$. Adoption of new technology leads to a lower marginal cost curve C'_1 with $-C'_1(e_i) < -C'_0(e_i)$ for all $e_i \leq e_i^{\max}$. The incentive for adoption is given by the firm's additional profit from switching to the new technology. Since we assume the market is competitive, there is no need to explicitly pay attention to the output market. Thus, we can focus on the compliance costs. Let us analyze the adoption incentives in the case of a performance standard and MBIs like taxes or tradable permits.

Let $\bar{e}$ denote the performance standard and let p denote the tax rate or a permit price, respectively. With performance standards, the incentive for adoption is simply given by the cost savings resulting from using new technology when firms are restricted to emit no more than $\bar{e}$ units of emissions. If buying and installing the new technology involves a fixed cost $F > 0$, the firm decides to adopt the new technology if and only if $C_0(\bar{e}) - C_1(\bar{e}) \geq F$. Instead, with MBIs, firms' cost minimization requires marginal abatement cost to equal the permit price of emission for both technologies. Thus, $-C'_0(e_0) = p$ and $-C'_1(e_1) = p$, which yields emission levels $e_0(p)$ and $e_1(p)$. Then, a firm adopts the new technology if and only if $C_0(e_0(p)) + pe_0(p) - C_1(e_1(p)) - pe_1(p) \geq F$.

Figure 9-3 illustrates the adoption incentives under performance standards and taxes, where T denotes the size of the tax. MC_0 is the marginal abatement cost of the conventional abatement technology while MC_1 is the marginal abatement cost of the new technology. Under performance standards, the firm enjoys a lower cost on the units of emissions it was abating

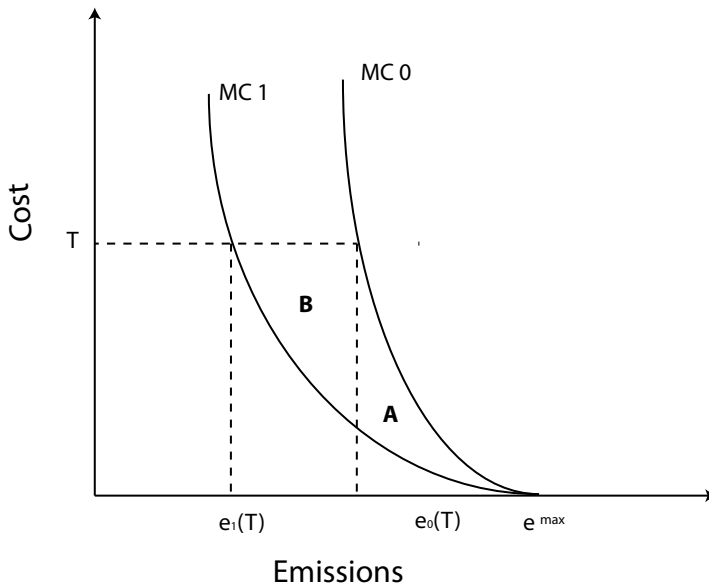


FIGURE 9-3 Single firm's investment incentive under performance standards and taxes.

Notes: MC_0 = marginal cost of abatement without the new technology; MC_1 = marginal cost of abatement using the new technology; T = optimal tax (at time t or 0); $\hat{e}_0(T)$ = optimal emissions without the technology, $\hat{e}_1(T)$ = optimal emissions using the new technology.

initially, before adoption. The savings on abatement are given by area A. Under taxes, the firm optimizes the level of abatement given its new technology. Since the abatement costs are lower, the firm will abate at a higher level. For each unit of added abatement the firm saves the difference between the tax it was paying initially and its new marginal abatement cost. Over the whole increase in abatement, this saving equals area B. Total cost savings are given then by area A plus B.

Instead of taxing emissions, the regulator may auction off or freely allocate tradable permits to the firms. Let p_0 denote the original price for permits (see Figure 9-4) before the new technology was available. Let us consider the case of a single firm that is small and it is not able to manipulate the permit price. If we assume that only this firm has access to the technology, its adoption incentive is the same as in the case of the tax. Nevertheless, if a considerable number of firms adopt the new technology the equilibrium price goes down. Let p_1 denote the new permit price. Therefore, each firm that has adopted the new technology has lower variable costs than in the situation where no other firm has adopted the new technology. The cost difference is equal to the area $A_1 + A_2 + B + C$ in Figure 9-4. However, this area is not equivalent to a single firm's incentive to adopt the new technology since we must distinguish between gains to firms due to private adoption decisions and gains that are enjoyed by all firms, regardless of their actions. Since the gain from the fall

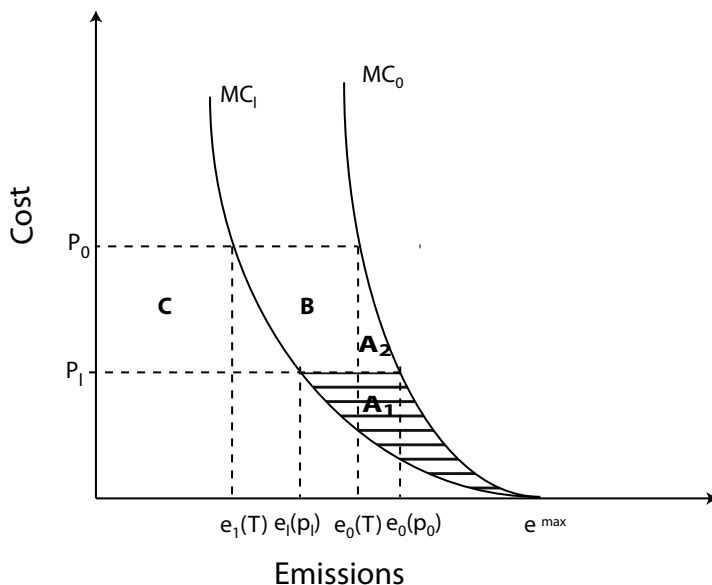


FIGURE 9-4 Single firm's investment incentive under tradable permits.

Notes: MC_0 = marginal cost of abatement without the new technology; MC_1 = marginal cost of abatement using the new technology; p_0 = permit price before the arrival of the new technology; p_1 = permit price once the new technology is diffused into the industry; $e_0(p_0)$ = optimal emissions without the technology, $e_1(p_1)$ = optimal emissions using the new technology.

in price (area $A_2 + B + C$) is enjoyed by every firm—regardless of its technology choice—it does not contribute to a firm's incentive to adopt. Instead, this incentive is given then by the shaded area in Figure 9-4, and it is lower than the adoption incentive under taxes.

Early research in the area (Downing and White 1986; Milliman and Prince 1989; Malueg 1989; Jung et al. 1996) suggested that freely allocated permits provide lower incentives to adopt the new technology than auctioned permits. The argument behind such a view is that firms postpone adoption in order to avoid the depreciation of the firms' permits due to the fall in the permit price. However, if firms are competitive, this depreciation affects every firm, regardless of the adoption decision. Therefore, they do not affect a firm's incentive to adopt. The allocation of permits does however matter when the adopter has market power in the permits market. In that case, an endowment effect reduces the incentives provided by freely allocated permits (see Montero 2002 and Coria 2009).

When analyzing innovation, adoption, and diffusion it is important to distinguish the possible timing of the policy response. So far we have assumed *ex ante* optimal policy. That is, the regulator makes a commitment to both the choice and the level of his policy instrument before firms invest. If the regulator has no incentive to change his behavior after firms have adopted, his

policy is time-consistent; thus the policy will remain at the initial level after the investment decisions.

By contrast, we talk about *ex post* optimal policy when the regulator adjusts the level of the policy instruments after firms have adopted. *Ex post* regulation is time consistent by definition. If the regulator can react optimally after firms have adopted and if the market is competitive, price regulations and quantity regulations (tradable permits) are almost equivalent. However, under imperfect competition the ranking of the different policy instruments is ambiguous since firms can under-invest or over-invest in order to influence the level of the policy under *ex post* regulation.

A couple of theoretical contributions have taken into account the uncertainty about technological progress. Phaneuf and Requate (2002) examine the effects of banking permits on the incentives to invest in advanced abatement technology if there is aggregate uncertainty about the abatement costs. Banking allows firms to postpone until the information about the abatement costs is revealed. However, the analysis shows that private and socially optimal responses to the resolution of uncertainty are not identical, which can lead to suboptimal levels of investment. Van Soest and Bulte (2001) analyze the case of technological adoption when technological advances are uncertain. Applying an option value approach, they show that even if it pays to adopt the new technology according to the criterion on net value comparisons, it may be not profitable if further improvements are likely to occur. Thus, firms postpone the adoption decisions until an even better technology is available. Using this approach, Van Soest (2005) compares energy taxes and energy use standards. Surprisingly, he finds that firms tend to postpone the adoption decisions when regulations are more stringent. Besides, there is no unambiguous ranking of the two instruments when it comes to stimulating early adoption of new technology.

Summarizing, it seems difficult to draw clear conclusions on which policy instrument dominates other policies because the ranking will depend on the behavior of the regulator, the market structure, and whether we pay attention to uncertainty. However, it is clear that market-based policies perform better than command-and-control policies, and that under myopic environmental policies, taxes provide stronger incentives to invest in both R&D and adoption of new technology than tradable permits.

Exceptionally little empirical analysis has examined the effects of the choice of policy instruments on technology innovation and adoption of abatement technologies, perhaps because of the paucity of available data (Jaffe et al. 2002). Econometric analyses of the lead and acid rain programs in the U.S. found evidence that tradable permits provide incentives for more efficient technology adoption decisions, as evidenced by the difference in adoption behavior between firms with low and high compliance costs (Keohane 2001; Kerr and Newell 2003). When it comes to energy efficiency technologies, firms have been shown to be more responsive to energy prices than to energy

nonprice regulatory constraints (Newell et al. 1999; Ohta and Griliches 1976; Pakes et al. 1993). However, empirical studies have also shown that the diffusion of energy-efficiency technologies is more sensitive to variation in adoption cost than to energy prices (Jaffe and Stavins 1995; Hassett and Metcalf 1995). Therefore, subsidizing the adoption costs partially might be a useful mechanism to speed the rate at which new technologies are diffused.

Fischer and Newell (2008) assess different policies for reducing carbon dioxide emissions and promoting innovation and diffusion of renewable energy. They find that due to knowledge spillovers, the optimal policy involves a portfolio of different instruments instead of single policies; an optimal portfolio of policies—including an emissions price and subsidies for technology R&D and learning—achieves emissions reductions at significantly lower costs.

Weber and Neuhoﬀ (2010) analyze the effects of firm-level innovation in carbon-abatement technologies on optimal cap-and-trade schemes with and without price controls and compare it to the cases of pure taxation and a simple emissions cap. In their model, an increase in the effectiveness of innovation lowers the optimal emissions cap and leads to relaxed price controls unless the slope of the marginal environmental damage curve is small. Due to the decrease in the emissions cap, innovation in abatement technologies can lead to a higher expected carbon price, so as to provide sufficient incentives for private R&D investments. Successful innovation shifts the trade-off between price and quantity-based instruments toward quantity.

Finally, another body of research has examined the effects on technology diffusion of “differential environmental standards” that require a much larger level of pollution abatement from new sources than for existing ones. Applying the most stringent controls on new sources raises their cost relative to existing sources. Hence, these regulations have lengthened the time before plants were retired, and since new sources are typically cleaner than older ones, they worsen pollution by encouraging firms to keep older and dirtier plants running (Stewart 1981; Gollop and Roberts 1983; Maloney and Brady 1988; McCubbins et al. 1989; Nelson et al. 1993). For instance, Nelson et al. 1993 study the relationship between environmental regulation and capital turnover in the electric industry in United States. Using a pooled database of 44 electric utilities over the 1969–1983 time period, they find that imposing more stringent controls on new sources did retard capital turnover by 24.6%. The impact of differential regulation appears to be significant for transportation as well (Chapter 16). Gruenspecht (1982) analyzed the effects of differential regulation on the rate of turnover in the automotive fleet in the United States; these regulations were equivalent to a setback of three to four years in the timetable for reducing emissions.

Growth and Inflation

Besides technological progress, many other variables in the economy change over time. The most obvious are overall income level and price level. The effects of economic growth and inflation differ by policy instrument. Inflation may be seen as only a change in the value of money, and as such, it would not warrant changes to quantitative instruments such as permits (whether tradable or not); however, all price-based instruments—particularly taxes but also subsidies, refunded charges, deposit–refund systems, liability bonds, and fines—would need to be adjusted.

The difficulty of making such updates also depends on the instrument. Tax changes typically have to pass through a legislative body, making them fairly complicated and somewhat unpredictable to modify. Administrative fees are simpler in this respect. Taxes and charges can be indexed to some measure of the price level, but this kind of indexing may be considered to fuel inflation and is not always popular with politicians because it is perceived as reducing their discretion. If inflation lowers inappropriate subsidies without the need for painful political fights, it may be an advantage. On the other hand, rapid inflation often is partly due to high budget deficits, making all subsidies unsuitable because they contribute to such deficits.

Economic growth (and population growth) requires more fundamental change, because such growth means that the curves for marginal cost of abatement and marginal cost of damage move; thus, the optimal level of pollution changes, too. Unless strong nonlinearities in the damage function provide a clear threshold value as a fairly stable goal, permit and tax levels would have to change as a result of economic growth. Usually, this process is slow; however, under some circumstances, local growth rates in particularly attractive or dynamic regions may be high, which would require rapid policy responses.

Supplemental Reading

Baumol and Oates 1988 (Chapter 8)	Jaffe et al. 2005
Coria 2009	Requate 2005a
Fischer and Newell 2008	Hart 2008
Hoel 1998	Weber and Neuhoﬀ 2010

Notes

- 1 Actual physical regulation may well be more subtle, and if regulators can correctly estimate marginal costs and adjust emission reductions or permits accordingly, then regulation may come close to the optimum (e.g., Brännlund et al. 1996). Individual regulation may offer advantages, particularly when damages are heterogeneous.

The risks and difficulties involved in this kind of differential standard setting are notorious because they provide a strong incentive for firms to misrepresent their true abatement cost functions (see Chapter 10). Some transport examples are presented in Part 4 and industrial examples in Chapter 20.

- 2 See Chapter 10 for more analysis of the choice between price-type instruments and regulation. Because the rate of emissions is the time derivative of the stock level, in order for the marginal damage of an emissions curve to be steep, damage would have to be a rapidly accelerating function of stock. Although this result is possible, it does not seem to be in line with the current estimate of the situation. However, for stocks that are on the verge of irreversible collapse, the implication may be a steep marginal damage (see Chapter 22).
- 3 One example is the U.S. sulfur emissions trading program, in which permits initially were expected to sell at \$300–\$700/ton but ended up selling at around \$100/ton; another example is the case of cleaner diesel fuels in Sweden (see Chapter 20). Comparing ex post and ex ante cost estimates for pollution control provides more empirical evidence (Harrington et al. 2000).

Role of Uncertainty and Information Asymmetry

In the previous chapter, we focused on the relative efficiency of policy instruments under different economic conditions. In this chapter, we focus on policy design under uncertainty and asymmetric information. In everyday language, the problems of adverse selection and moral hazard are usually labeled “uncertainty.” However, in more technical parlance, it is customary to reserve the term uncertainty for stochasticity or randomness of the environment. It must be distinguished from a situation where one party has more or better information on some aspect of the economic relationship than the other. This is the case under asymmetric information, under which the information gap will be exploited if by doing so the better-informed party can achieve some advantage.

One of the earliest and most famous articles in environmental economics focuses on formalizing the design of policy instruments under conditions of uncertainty (Weitzman 1974). It focuses on the choice between regulation through permits and regulation through taxation. This choice turns out to hinge on the slope of abatement and cost curves. Sometimes, neither permits nor taxes can be used, and the lack of information as well as the asymmetric information problems can make policy design quite complicated.

First, we briefly present some general analyses of uncertainty and asymmetric information and then several cases in which the availability or provision of information to some agents can facilitate policy implementation. Examples include the provision of aggregate or ex post information or of information to peers and third parties.

Uncertainty in Abatement and Damage Costs (Price vs. Quantity)

The seminal Weitzman article, titled *Prices vs. Quantities*, concerns the choice between a price (P) instrument, which is a tax or charge, and a quantity (Q) instrument, which may be thought of as a tradable permit system (Weitzman

1974). The regulator is assumed not to have full information on abatement costs. When there is certainty, the two instruments can be seen as equivalent; both lead to the optimal outcome.

If there is uncertainty about the benefits of environmental cleanup, there will be uncertainty about the appropriate target level for the charge or regulation, but this uncertainty does not affect the comparison between charges and permits. The two instruments give the same (uncertain) result because the results of the model are determined by the firms' abatement costs. However, if these abatement costs are uncertain, then the expected outcome will be different with the different instruments¹. If the marginal cost of abatement is mistakenly overestimated, then the regulation would be less stringent than the optimal one, whereas if charges were chosen, they would be set too high—leading, implicitly, to an excessively stringent abatement.

As shown in Figure 10-1, the price and quantity instruments do not just lead to a symmetric over- or underregulation. With overestimated marginal cost (dashed lines), the regulator will aim for OE instead of O, and the tax level (P) will be too high, leading to overabatement (AP). On the other hand, regulation will be too lax (at AQ), but the size of the economic welfare cost caused (as measured by deadweight losses, i.e., welfare losses due to distortions in consumption choices caused by distorted prices, for instance) depends on the relative slope of the marginal abatement and cost curves. If errors in judgment are assumed and the sum of expected consumer and producer losses are minimized, then the expected losses turn out to be different with P- and Q-instruments, as shown by the relative size of the deadweight triangles; the assumption that the curves are linear makes for a simple algebraic analysis (Weitzman 1974). If the marginal benefits of abatement are flat and the marginal costs of abatement steep (see Figure 10-1), then one can predict with reasonable precision the “price” for pollution, and the loss due to taxation will be small. The exact regulatory level is difficult to give with precision, and the risk of large costs is considerable with the Q-type instruments. The opposite applies when the benefit of abatement (or pollution damage) is steep and the marginal cost of abatement flat: the deadweight loss due to an excessively high pollution tax would be large, whereas the error caused by choosing a given target abatement level is small. Thus, a P-instrument should be used if the cost of abatement curve is steeper than the damage curve, and a Q-instrument should be used when the abatement cost curve is flatter than the damage curve.

Since Weitzman's original analysis (1974), numerous articles have explored aspects of the P-versus-Q issue. For example, Laffont (1977) and Watson and Ridker (1984) compare taxes and permits in the case of nonlinear functions and more realistic error terms; however, the results remain the same.

One intriguing suggestion is to introduce hybrid instruments that allow polluters or users a choice between buying permits and paying a charge. For instance, an environmental protection agency might issue several permits corresponding to its target pollution level but allow additional permits to be

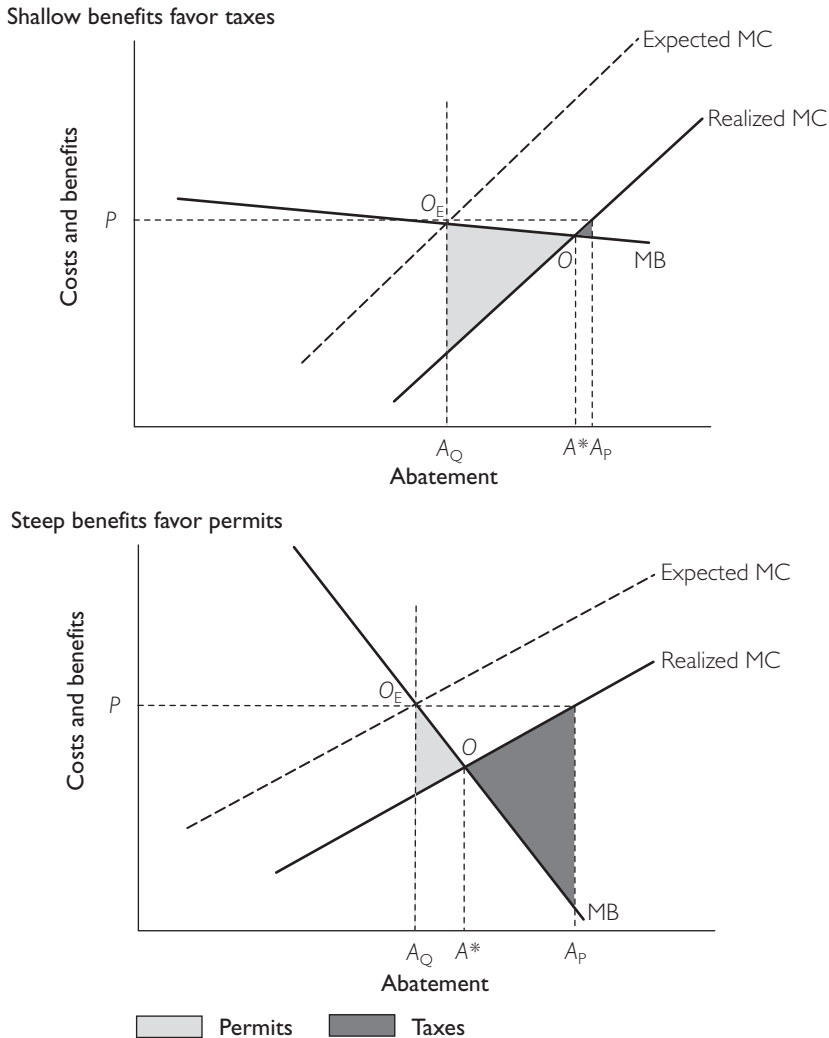


FIGURE 10-1 Deadweight Loss of Taxes versus Permits

Notes: P = optimal tax level; O = target for optimal policy; O_E = target for policy under expected costs; MC = marginal costs; MB = marginal benefits; A_Q = abatement under regulation; A^* = optimal abatement; and A_P = abatement under a tax.

bought at a certain (but high) rate as a “safety valve” for companies that might otherwise face unbearably high abatement costs. The combination of the two policies in a hybrid policy should reduce the risk of large errors. Many hybrid policies are potentially available, including very high fees for noncompliance. One suggested hybrid includes permits and side payments (taxes and subsidies) to better emulate a nonlinear (increasing) function such as the environmental damage of increasing pollution levels (Roberts and Spence 1976).

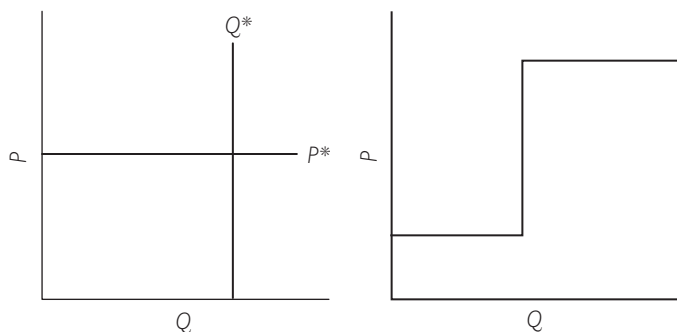


FIGURE 10-2 Conventional (left) and Step-Shaped Hybrid Policies (right)

Notes: P = optimal tax level; Q = quantity; P^* = optimal tax; and Q^* = optimal quantity regulation.

Another suggestion is that the regulator should simply adjust the number of permits by open-market operations (Collinge and Oates 1980).

A hybrid mechanism could combine a limited number of permits that are freely allocated with a menu of call options for additional permits (Requate and Unold 2001). Different sets of permits would have different prices, thus allowing regulators to reflect the rising costs of environmental damage. Diagrammatically, these policies can be thought of as step-shaped functions as opposed to the vertical and horizontal functions that symbolize Q - and P -instruments, respectively (see Figure 10-2). This kind of policy might have a low fee (or free or cheap permits) up to a certain level and then high charges or fines (or more expensive permits) in excess of that level.

In practice, different emitters of a particular pollutant are sometimes subjected to different control mechanisms. For example, the EU Emission Trading System (ETS) covers only 46% of the European CO_2 emissions (see Chapter 6), and the rest is confronted by taxes. According to Mandell (2008), this division of the economy in two sectors—subjecting one sector to cap-and-trade and the other to an emission tax—is optimal from an efficiency point of view. The reason is that under a mixed control mechanism, the aggregate emissions volume will be closer to the efficient volume regardless of the final realization of the benefits and costs.

In almost all the literature, the word “uncertainty” is taken to describe the realization of an event for which the true probability distribution is known. Thus, the expected utility framework can be used. On the other hand, pure or Knightian uncertainty, describes the realization of an event where the space of outcomes is known but the decisionmaker is unable to assign probabilities. Stranlund and Ben-Haim (2008) analyze the choice between priced- vs. quantity-based regulations under Knightian uncertainty (see also Box 10-1); the main result is that the same criterion involving the relative slopes of the marginal benefit and cost functions determines whether price or quantity control is more robust to unstructured uncertainty.

BOX 10-1 Knightian Uncertainty, Climate Change, and the Precautionary Principle.

Global environmental risks such as climate change are events with major widespread and possibly irreversible consequences. At the same time, an essential component in climate modeling is the scientific uncertainty regarding climate sensitivity and the unknown distribution of climate sensitivity outcomes. Hence, since expected utility theory requires that objective or subjective probabilities can be assigned to each outcome, such an approach might fail in capturing the reasonings behind the precautionary principle, which states that when threats of serious or irreversible damage are present, lack of full scientific certainty shall not be used as a reason for postponing cost-effective measures to prevent environmental “degradation.”

Instead, Hennlock (2009) demonstrated that Knightian uncertainty makes policy more stringent and speeds up the timing of policy response to unpredicted increases in observed climate data. Hennlock also found that introducing deeper uncertainty in a game-theoretic setting alters the incentives to international cooperation in climate change policy (see Chapter 14). Without uncertainty, countries are mainly interested in their own welfare. However, introducing uncertainty may modify these incentives. The intuitive explanation is that in such a setting climate agreements acquire a second function: they provide insurance that diversifies ambiguity over all participating countries.

Hazardous emissions with distinct threshold values warrant a Q-type instrument, such as a ban or a regulation. The same may apply to the management of a natural resource in danger of extinction. However, steep abatement costs (and flat damages) favor the P-instrument. Steep abatement costs may be due to cost heterogeneity (see Chapter 9), which reinforces the advantages of price-type mechanisms. It is not easy to generalize about which economic conditions give rise to steep or flat cost curves. The application concerned as well as the character of the economy determine the shape of the curve. It is sometimes assumed that countries with low income and little previous pollution control have low abatement costs (because they are at the “beginning” of the abatement curve). However, such economies often have limited access to credit, technology, and skilled technical staff—factors that may make abatement difficult. They also typically have high unemployment rates and fear the flight of capital to other countries and maybe social unrest if unemployment increases. Consequently, the abatement curve can be quite steep.

Costs of abatement change rapidly with new technology and with the restructuring of industry. In many transitional (and some developing) countries that have old, polluting plants, abatement could be so expensive, at least in the short term, that a given environmental standard might bankrupt a company. In an economy with high unemployment and extensive poverty, the bankruptcy of a single firm may be such an undesirable outcome that it will be resisted at all cost, which may help explain the observed preference for low environmental fees (with revenue recycling), which place a small and predictable burden on the enterprise, thereby minimizing the risk of bankruptcy while providing some incentive for gradual cleanup.

Asymmetric information Concerning Type of Polluter or User

In a bilateral relationship in which one party contracts another to carry out some type of action, the contractor is usually referred to as the principal while the contractee is referred to as the agent. Both principal and agent could be individuals, institutions, or organizations. The economics of information is centered on a fundamental aspect of the contractual relationship: the effects on the final outcome of an informational advantage of one of the parties. Such information asymmetry leads to different types of models. When the agent has relevant private information before the contract is signed we have an adverse selection problem. When the information asymmetry arises after the contract has been signed we have a moral hazard problem.

The adverse selection problem is common in the insurance literature. The insurance company knows that some people are more prone to get sick than others, and it is assumed that the individuals know their own risks but that the company cannot readily find out². If an insurance company asks, "Are you likely to get sick?" then an applicant has no incentive to tell the truth. However, the company can indirectly get the answer by designing two or more plans, one with different combinations of premium and copayment or deductible. A low premium with a high copayment or deductible will be more attractive to individuals who have low risks, and a high premium with a low copayment or deductible will be more attractive to individuals who have high risks. Thus, individual applicants reveal their risk types in choosing their policies.

Note that this design is still not optimal in the sense that ordinary individuals cannot buy full insurance at a price that corresponds to their true risk; neither high-risk nor low-risk individuals get the optimal insurance they would have been able to get without asymmetric information. Therefore, some low-risk individuals may choose not to buy insurance at all because of adverse selection, making the premiums more expensive for those who do. A similar principle determines the structure of the contracts discussed in Chapter 2: an employer cannot judge the types of workers and thus may offer a low fixed wage; a landowner cannot judge the types of tenants and may offer some mix of wage and sharecropping. Either way, wages are not equated to marginal productivity, and incentives for effort are not the same as in a risk-free world.

A similar principle determines the structure of policy toward different kinds of polluters. In a situation where pollutants are not mixed and abatement costs vary, individual tax rates may be needed (see Chapter 9). However, if regulators do not know the abatement costs of each polluter, polluters will have an incentive to claim low costs in order to get low tax rates. To get polluters to voluntarily reveal that they have high costs of abatement, two-part contracts can be designed that combine mechanisms—for example, individual emission fees with a subsidy for certain kinds of abatement technology.

Consider an agent (individual or company) that has private information about its type (described by the variable $\tau = \{\tau_i \mid i = 1, \dots, n\}$). This type could

be productivity, risk aversion, abatement costs, or the emissions factor of an unobservable pollutant. For an emissions factor, the principal (an environmental protection agency) cannot observe the type directly but is assumed to know the set of possible types as well as their distribution. For any given individual, there is an estimated probability (Ω_τ) that the agent's type is τ . Any other information that regulators have (e.g., the production function and emissions for each type) can be used to design policies that encourage firms to cooperate. One policy will be the most attractive to each polluter, and the principal has sufficient general information about technologies to calculate the optimal alleged type τ for each actual type τ . In this sense, the choice of contract reveals the type of the agent (see Box 10-2 for a formal definition).

A concrete and detailed example of the revelation principle as pertains to the design of contracts that internalize the use of scarce water is presented in Chapter 23, which deals with the design of contracts when metering of

BOX 10-2 The Revelation Principle

Formally, the agent selects an action a from a set $A = \{a \mid a_{\min} \leq a \leq a_{\max}\}$. The outcome (pollution) can be any one of M outcomes q_j . The probability of outcome j depends on action a and the type (τ), as in $\Phi_j(a, \tau)$; a itself depends on τ . The probability is assumed common knowledge. A contract function is a set of contingent payments $\omega = \{\omega_j\}$ from the principal to each agent for each outcome q_j . The utility (U) of the principal is a probability-weighted average of profits $u(q_j - \omega_j)$ based on an estimate of the probability distribution over types (Ω).

$$U(\omega, a) = \sum_j \sum_i \Phi_j[a(\tau), \tau] \Omega_\tau u(q_j - \omega_j) \quad (10-1)$$

It is not assumed that the agent will be truthful but that the agent will assert a type τ' corresponding to the contract found most profitable. The utility of an agent type τ who chooses a contract made for type τ' is

$$V\{\omega(\tau'), a \mid \tau\} = \sum_j \Phi_j[a(\tau), \tau] v[\omega_j(\tau'), a \mid \tau] \quad (10-2)$$

where v is utility in given outcome j . This is thus the payoff function the agent maximizes by reporting a type τ' . However, the principal can calculate the optimal alleged type τ' for each actual type τ . The principal maximizes Equation 10-1 given the following conditions:

$$V[\omega(\tau), a(\tau) \mid \tau] \geq V^*(\tau) \text{ (for all } \tau) \quad (10-3)$$

$$[\tau, a(\tau)] \in \operatorname{argmax}_{(\tau', a')} V[\omega(\tau'), a' \mid \tau] \text{ (for all } \tau) \quad (10-4)$$

Condition 10-3 is the reservation constraint that the agent utility must equal at least some baseline utility level V^* . Condition 10-4 is the incentive compatibility constraint that the contracts must be designed so agents find it profitable to disclose their true type τ rather than another type τ' .

water is not possible. There, the only information regulators have is the range of technologies used (i.e., the water demand functions for distinct types of farmers). Regulators can write self-revealing contracts that combine an offer to buy a certain volume of harvest with type-specific taxes on water. With similar information on the emissions coefficients of different types of producers, an environmental protection agency could design contracts with allowable production volume and fixed fees that would make polluters self-select, thereby allowing the environmental protection agency to steer production to minimize pollution. For example, French water agencies have designed contracts that give firms incentive to treat their water over and above the levels motivated by the current tax (which is too low to provide adequate incentive) (A. Thomas 1995). This kind of principle might be useful in regulating informal enterprises or other cases in which direct monitoring is impossible.

Markets to allocate pollution rights play an important role in current efforts to control pollution efficiently (see Chapter 6 and Chapter 20). Nevertheless, if the regulator knows little about firms' pollution abatement costs, he will be unable to establish the efficient level of pollution. Although the previous mechanisms might help to induce firms to reveal their type, the problem becomes more complicated if we recognize the strategic incentives of firms to exercise market power (i.e., to influence the prices they pay for emission permits). Montero (2008) proposes an auction mechanism that produces the efficient allocation of pollution as an equilibrium outcome in a Nash equilibrium. His mechanism builds upon a conventional uniform-price sealed-bid auction, but it has two additional features. First, since the regulator is clueless about the efficient number of permits to be allocated, the number of permits is endogenous to the demand schedules submitted by firms. Second, part of the auction revenues are returned to firms, not as lump-sum transfers, but in a way that incentivizes firms to bid truthfully. The two ingredients—endogenous supply and rebates—enter into the uniform price format in such a way that the resulting auction mechanism is both *ex post* efficient and a dominant strategy for the firms to reveal their type. Hence, the supply of permits reflects the cost to society of allocating the permits to firms, while the rebates are such that the total payments for permits of each firm are exactly equal to the “damage” the firm exerts upon all the other agents.

Asymmetric Information Concerning Polluter or User Behavior

In the previous section, the polluters held private information before the relationship started, but the regulator could verify the polluters' behavior. A somewhat similar class of problems arises when the payoff to the principal depends on the behaviour of the agent, but that behavior cannot be observed or monitored inexpensively. The essence of such a moral hazard problem is not hidden information but hidden action, although both could occur at the same time.

With hidden action, the challenge is to design a contract that offers incentives for the desirable level of performance of the agents. In general, this is done by making the agent's payment a function of the outcome as in the sharecropping models discussed in Chapter 2.

In the analysis of moral hazard, it is assumed that regulators know the agent's indirect utility function (V), which makes it possible for them to anticipate the polluter actions and thus work out the optimal strategy (see Box 10-3).

Unfortunately, in many real-world situations, regulators must design instruments with little knowledge. They may assume that firms maximize profit and that households maximize utility, but they do not know the production or utility functions, and then the lack of information is more general. However, when some useful information is available, regulators should use the information that is correlated with the desired outcomes. In the following sections, we present some of these special cases: when ex post (actual) information is available, when only aggregate information is available, and when information is available only to peers or third parties.

BOX 10-3 Moral Hazard

In a nonstochastic, single-agent moral hazard problem, the agent chooses an (unobservable) action a from the set A , giving M outcomes (as in Box 10-2). The probability (Φ) then depends only on the action a . A contract between the principal and the agent is a pay-off schedule (Ω) equal to a pay-off or wage $\{\omega_j \mid j = 1, \dots, M\}$ for each outcome (j). The expected utility (U and V) of the principal and agent are then

$$U(\Omega, a) = \sum_j \Phi_j(a) u(q_j - \omega_j) \quad (10-5)$$

$$V(\Omega, a) = \sum_j \Phi_j(a) v(\omega_j, a) \quad (10-6)$$

where ω is a set of contingent payments and u is the agent utility for a given outcome. Again, the principal designs the contract to maximize expected utility U subject to the following conditions:

$$V(\Omega, a) \geq V^* \quad (10-7)$$

$$a \in \operatorname{argmax}_{(a')} V(\Omega, a') \quad (10-8)$$

where Condition 10-7 is the individual rationality constraint and Condition 10-8 the incentive compatibility constraint. (If the action were observable, only Condition 10-8 would be needed, but then there is no moral hazard problem.) Assuming the agent chooses a to maximize the expected utility and assuming a unique interior solution, Condition 10-8 can be replaced by the following first-order condition (Laffont 1989a, 1989b):

$$\frac{\partial V(\Omega, a)}{\partial a} = 0. \quad (10-9)$$

Ex Post Availability of Information

In many cases, direct monitoring is almost impossible, but ex post verification of abatement may be possible. One area in which many such examples are found is in the inappropriate disposal of hazardous waste, in which small quantities of highly toxic materials are very difficult to monitor. Monitoring the crushed rock or tailings left by a mining company or the total volume of trash from companies or households is relatively easy. However, monitoring the disposal of a few grams of arsenic, cadmium, or mercury or even residual quantities of pesticides and household chemicals can be difficult.

It is hard to envisage the kind of monitoring that would enable an environmental protection agency to trace such small amounts of individual emissions occurring anywhere at any point in time. Even for large companies that monitor their main effluents, it is difficult to devise a system that can detect a brief, maybe momentary, dumping of some hazardous chemical—which could perhaps even be timed to coincide with scheduled repairs or with breakdowns of monitoring equipment. Yet such releases of some chemicals fulfill the dual criteria of potential harm and incentive: they can cause grave damage to the environment, and their illicit release could result in large cost savings for a firm (or an individual).

One solution that is sometimes appropriate is the use of mandatory equipment (e.g., filters, catalytic converters, and storage ponds) designed to abate emissions of the pollutant in a way that is at least satisfactory (if not necessarily optimal) and is highly visible. The importance of visibility is to facilitate monitoring. The potential disadvantages of this solution include the usual problems related to mandatory technology: mandatory equipment does not necessarily lead to optimal results, and incentives to develop better technology do not materialize; and equipment may be operated nonoptimally—automobiles might have effectively nonoperational catalytic converters if fuel is poor, or factories may have improperly used or maintained filters and storage ponds, for example.

Another class of instruments that goes more directly to the heart of the issue builds on a deposit–refund concept. Shifting the burden of proof to the agents, the environmental protection agency charges potential polluters as if they were polluting but then refunds the charge if they can prove compliance by handing in the physical residues (or by following an appropriate procedure for certification of disposal). Deposit–refund schemes are used mainly for bottles and cans, but one Swedish example keeps scrap cars from being dumped in the woods. All car buyers pay a deposit that is refunded when the car is turned in for demolition. Potential applications extend further. For example, any emissions of mercury or sulfur (which cannot be created or destroyed in the process of production) must come from the inputs the firm acquires. Therefore, regulators could tax the purchase of the input and then pay a refund for its return. (This policy does not work for synthetic toxins such as chlorinated organic compounds or pollutants like NO_x , which are manufactured and can be destroyed.)

Various pitfalls may be associated with setting the appropriate deposit. In principle, the deposit should be set at the Pigovian level to reflect the potential damage from inappropriate disposal (Equation 7-1 in Chapter 7). However, these damage costs may be very high, and the deposit cannot be higher than production costs—otherwise, there might be a risk that inputs would be produced or imported for the sole purpose of collecting the refund³. It might not be necessary for the deposit to be at the Pigovian level if the costs of abatement (i.e., collection or appropriate disposal) are limited. The fee only needs to be high enough to encourage collection and safe disposal. For households, it means overcoming the inconvenience costs of storage and disposal. (Some empirical examples of sulfur taxation and waste management fees in Sweden are presented in Chapters 20 and 24.)

Discussion on deposit–refund schemes tends to focus on beverage containers, but deposit–refund schemes are part of a larger class of two-tiered price schemes that can be used in a much broader set of circumstances. Similar concepts are used to entice motorists to buy less environmentally dangerous vehicles and to have cars inspected (see Chapters 16, 18, and 19). Mandatory programs such as vehicle inspection and maintenance are difficult to enforce; owners of the oldest and most polluting vehicles have an incentive to evade control. One possible response is to mimic the refund mechanism by making it attractive for vehicle owners to have cars inspected (or to buy cleaner vehicles)—for instance, by raising annual registration fees but paying rebates for vehicles that somehow reveal themselves as cleaner through certification, on-board diagnostics, or inspection. Similarly, owners could receive rebates for vehicles that demonstrate (through global positioning or similar systems) that they are not driven in urban areas where environmental damage is costly.

Availability of Aggregate Information (Nonpoint-Source Pollution)

Diffuse pollution, referred to as nonpoint or nonpoint-source pollution (NPSP), is a category of cases for which individual monitoring is lacking not because the levels of emissions or effluents are low, but because emissions are spread over a large area. Usually, monitoring the ambient levels of pollution that depend on emissions from a group of sources is possible. The partial information that this monitoring provides makes the use of policy instruments such as ambient charges possible. However, under some circumstances, other mechanisms that build on (and foster) cooperation between the polluters are preferable.

NPSP includes many important categories of environmental problems. A prime example is water pollution from the agricultural runoff of fertilizers and pesticides, which is extremely costly to monitor by its nature (i.e., diffusely spread over large geographical areas, unevenly and unpredictably distributed over time). Agricultural runoff causes the eutrophication (accumulation of

excessive biological matter) of streams and lakes, pesticide accumulation in rivers and coastal waters, and soil erosion—the consequences of which include the death of fish, other water species, or ecosystems (including coral reefs). Moreover, toxic chemicals accumulate in the food chain, and dams or power stations must cope with obstructions due to siltation. All of these consequences have negative effects on fisheries, recreation, tourism, health, and many other sectors. Furthermore, NPSP issues are interesting at a theoretical level because most environmental issues have some degree of NPSP character (i.e., some degree of monitoring cost), and analyses developed for the pure NPSP cases may throw additional light on other cases of industrial or transport pollution where monitoring is difficult.

Consider a simple setting in which n polluters share a watershed and total pollution is to be regulated. If “perfect mixing” is assumed, then all the transfer coefficients are identical (and equal to 1), so the only concern is aggregate emissions (E), which are equal to $\sum e_i$. If the equilibrium charge level is T , then an ambient charge paid by every polluter (TE) would have the right incentive properties at the margin, because $TE = Te_i + Te_{-i}$, where Te_i is a standard Pigovian charge on emissions for agent i and Te_{-i} is a fixed transfer payment from the viewpoint of agent i (e_{-i} is the sum of all e except for agent i). In reality such a tax would have numerous problems. It assumes that E is known, even though in reality, the regulator has only imprecise estimates. The tax level is also far greater than the damage caused by each polluter. With serious externalities and large groups (large n), the payment TE may even exceed what individual agents earn and thus what they can pay. Furthermore, an equal fee would be unfair for small producers. This scheme brings in total charge revenues (nTE), which is n times total damages, which may pose serious problems vis-à-vis the right of local communities to raise such taxes. The fact that the revenues are so large provides the option of modifying the instrument in some way to refund the revenues.

Several increasingly sophisticated solutions have been suggested for dealing with policy design in situations with aggregate information. An early important contributor to address the NPSP problem suggests two possible reasons for the inability to infer behavior from observed outcomes (Segerson 1988). First, given any level of abatement, the effects on environmental quality are uncertain because of stochastic (random) variation. Second, the emissions of several polluters contribute to ambient levels, and only combined effects are observable. Because of the presence of uncertainty and monitoring difficulties in an NPSP situation, mechanisms that provide incentives for compliance must be used instead of direct regulation of each polluter’s discharge. In essence, this suggested solution is a combination of the ambient charge and a penalty, which leads to a first-best allocation when polluters are risk-neutral and follow Cournot–Nash (self-interested) behavior. If the ambient pollution level is less than the predetermined standard, then firms receive a positive payment. If the ambient pollution level exceeds the predetermined standard,

then each polluter has to pay a fee that is proportional to the level of ambient pollution, plus a penalty for exceeding the ambient pollution standard.

If monitoring is not impossible but just costly, then it is assumed that the regulator will monitor sometimes, but not always. The expected cost to the firm would be ΨT , where Ψ is the probability of detection and T is the fee charged for noncompliance (theoretically, T is based on damages, but because damages are difficult to calculate, the value is often based on a rule of thumb or an estimate of how much the firm saves through noncompliance). If Ψ is small, then the expected value of the fee is likely to be insufficient to motivate compliance, but increasing the fee might encourage compliance. The fee level that ensures compliance might be too high to be feasible, but assuming risk aversion, the fee can be set lower and still have an incentive effect that would be on a par with a certain fee of T . Several mechanisms have been suggested along these lines to enforce NPSP regulation⁴.

Availability of Information to Peers or Third Parties Only

If monitoring is so difficult that regulators have no information, then they have little alternative but to pay for abatement, which is an example of the direct provision of public goods (possibly with voluntary agreements or cost sharing by the whole industry concerned). Sometimes information may be available only to the polluters' peers. In such cases, collaborative policy designs such as common property resource management may be appropriate (see Chapter 27).

Risk, Insurance, and Environmental Policy

When economic outcomes depend largely on the random variation of nature, insurance (or other mechanisms to even out the revenue flow, such as savings) will generally be required. In the presence of adverse selection and moral hazard, the demand for insurance will be at least partly unsatisfied. For this and other reasons, many developing countries have poorly developed markets for insurance, banking, and related services—conditions which may be relevant for the choice of environmental policy instruments.

Sometimes the risks are private, but the absence of insurance leads to public risk in the form of pollution. For example, individual farmers bear the private risk of whether their harvests will be reduced by pests, but they create a public risk by using excessive amounts of agricultural pesticides (see Chapter 26). Overstocking of cattle is another mechanism through which pastoralists typically attempt to compensate for the lack of formal savings institutions. Unfortunately, this kind of system often leads to overgrazing, which has negative consequences for the environment. Such situations might be addressed by charges on or regulation of pesticides or number of cattle. However, given the potential gravity of the welfare losses, these policies generally are neither

politically feasible nor desirable. Instead, the appropriate response may be to provide or facilitate the provision of insurance or banking services (or the institutions that create conditions for the market to supply these services).

At the other end of the scale are isolated large-scale events that lead to private risks for all or many citizens. Such events include accidents at nuclear power plants or chemical plants, such as the Union Carbide plant in Bhopal, India, which leaked toxic gas into the local community in 1984, killing or injuring thousands of people. Because of steep damage curves, direct regulation is generally the most appropriate and commonly used policy. Technical complexity and the inherent difficulty of dealing with small probabilities add extra layers of difficulty. The magnitude of the worst possible event may be much larger than the net worth of the responsible firm; therefore, various arrangements may be considered, such as strict liability, mandatory insurance (at specified levels), or the provision of liability bonds (which are essentially set-asides of money to be used in case the company defaults on its obligations or goes bankrupt). Attention may need to be focused on the limited liability inherent in company law to make parent companies, stockholders, creditors, and other agents share the risk. The provision of public insurance, whereby the government implicitly or explicitly assumes risks to limit an industry's liability (e.g., the Price-Anderson Act, which limits the liability of the U.S. nuclear power industry), may be an incentive for business but may also worsen moral hazard and thereby require even more regulation.

In both risk scenarios, the provision of information to the producers, communities, banks, insurance firms, customers, and society is an important by-product of insurance provision. By requiring and providing insurance, the market itself is forced to calculate, disclose, and provide information about the risks in a form that will be useful to several parties in society and help them make rational decisions. A somewhat different approach to risk and uncertainty is for firms to adopt the technology recommended by an environmental protection agency or similar authority explicitly, to protect themselves against criticism (or penalty) from the same agency in the event of any incident or accident. This mechanism gives authorities considerable leverage over technology but reduces their ability to hold the firms accountable as long as they follow the technical prescriptions. This approach turns an environmental protection agency into a clearinghouse for technology, a development that has happened in some areas of the United States.

Availability of Information to Consumers

Sometimes, the relevant primary decisionmakers on environmental issues may be not government regulators but individual consumers⁵. For instance, government regulation of the level of pesticide residues in and on food may be fairly tight, but consumers can still make an additional trade-off between price and quality by choosing certified organically grown produce to suit

their tastes and budgets. Such choices cannot be made without information, and the assumption of “full information” is not very appropriate here. The natural science and medical facts concerning pesticides and other toxins are so complex that few laypeople can grasp them, even if they have the information that usually is available only to the producers. In such cases, information might be considered a public good, and government agencies should provide the institutions to aid its dissemination and interpretation.

In recent years, several technologies have been developed that make the provision of and access to information as well as its processing, assimilation, and comprehension easier (e.g., remote sensing, satellite photography and mapping, genetic and other biotechnology, advances in computing, and the ubiquitous Internet). The difficulty of transmitting information about quality is a problem for consumers as well as producers and applies not only to environmental attributes (such as clean products or processes) but also to all other quality attributes that are subtle or difficult to detect. It is sometimes referred to as “the problem of the lemons.”

If inferior goods (“lemons”) are difficult to distinguish from high-quality products, then consumers have little incentive to buy high-quality products and probably will avoid them because they are more expensive. Good products are nearly always expensive, but a high price is no guarantee of quality because inferior products (which are indistinguishable) may also sell for a high price. The result may be an undesirable equilibrium in which only low-quality products are sold. If there is some way to distinguish the products through signaling—through warranties, information, or labeling—then the high-quality producers typically will try to use such techniques. For environmental attributes, the issue is complicated by the fact that the “quality” attribute is not private (such as product durability) but public. This feature makes it problematic for the seller to issue warranties and may be a motivation for public investment in and support for environmental labeling schemes.

Information provision is expensive and not necessarily wholly positive. For example, providing full information about dangerous chemicals may have the desired effect, making consumers wary of and cautious about the products. However, providing detailed information about less dangerous chemicals can have a similar effect, making consumers wary about those chemicals, too. How are consumers supposed to judge the relative dangers of 10 grams of zinc, 50 grams of an enzyme, or 1 nanogram of dioxin when emitted into different recipients under different conditions? A layperson, who does not know which chemicals and interactions are truly problematic, may be frightened by the mere sight of a chemical formula. There is thus some risk that information provision, particularly of raw data, can create a “scare,” but the advantage of this approach is that openness is encouraged and competing interpretations can argue in public. One might compare with the general dilemma of a democracy in which the electorate has to make many complicated choices they do not have full information about (“If we can choose governments, we should be able to

choose laundry detergents!”). One alternative to full information disclosure is the provision of processed information, but then the problem becomes one of determining the party to interpret the information. The solution is far from obvious, and the choice of policy instrument and the optimal degree of decision decentralization probably depend on the complexity of the issue at hand.

When monitoring is difficult and resources for information monitoring and processing are limited, the first priority may be to elicit information. The likelihood of receiving accurate information can be increased by standardizing the data-collection method (i.e., specifying the nature of the information to be gathered as well as collection or processing procedures, such as life-cycle analysis) and by making the penalties for falsifying information sufficiently large. The Toxics Release Inventory (TRI) is a large, structured collection of raw data about the use of chemicals in the United States (see Chapter 8). Organizations such as Scorecard use these data to provide maps and other easily accessible information that allow laypeople to make their own comparisons and informed decisions. Similarly, various national programs that promote environmentally friendly practices by awarding “green labels” to products (e.g., the United States’ Green Seal, Canada’s Environmental Choice, the Nordic Council’s White Swan, Germany’s Blue Angel, and Japan’s Ecomark) make an effort to interpret information for consumers (see Chapters 20, 21, and 25).

Information disclosure can be voluntary or mandatory. Farmers who use organic methods traditionally have sought organic certification voluntarily, because organic certification is considered to convey added value. (In contrast, no requirements are placed on conventional farms to list pesticides used.) On the other hand, “right-to-know” mechanisms (such as the TRI) require all firms to provide information about emissions.

The crucial issue for understanding labeling schemes is to understand the nature of the credence good, for which the value is derived from beliefs rather than actual experience (Nadai 1999). The technical issues involved are so complicated that the consumer cannot evaluate a green label, even after purchase of the good. The credibility of the issuing parties is thus of paramount importance. Clearly, consensus among all involved parties (industry, regulators, and possibly consumers or environmental organizations) is a good way to build credibility. As a result, industry is faced with an important strategic decision: whether to collaborate and influence the design of the criteria to arrive at a labeling scheme that is credible (and thus has marketing value), or to try to kill off the whole idea by opposing and discrediting labeling schemes in general.

How industries handle labeling schemes depends on factors such as the ease of satisfying criteria, the degree of consumer awareness and interest, and the ease of collaboration (collusion) within the industry. Documented approaches include the following (Nadai 1999):

- **Industry cooperation**—In the paint and varnishes industry, the main concern is to reduce volatile organic compounds (VOCs). Wanting to

maintain the distinction between water-based and other paints, the industry believed that it could best maintain profits through a collaboration that allowed influence over criteria in a way that would suit its interests concerning future technological developments.

- **Industry confrontation**—Manufacturers of hairsprays considered that the labeling requirements were difficult to achieve and not demanded by consumers.
- **Strategic groups**—Industry might split into strategic groups in which one or more collaborate and others do not, as in the laundry detergent industry (see Chapter 24).

Some degree of monitoring is vital, and in most cases is done by the regulated entity itself. When the mandatory reports submitted to environmental protection agencies are also publicly available, transparency is greatly enhanced, and the reports can be used by private enforcers as a basis for raising noncompliance claims. The reports also act as a check on not only the polluter but also the regulator. This element of discipline is emphasized in the design of the PROPER scheme in Indonesia (see Chapter 21) and public interest litigation in India. It is most effective in countries where the legal system encourages litigation, such as the United States. Management styles vary by region. For instance, North American industries appear to be more negative toward labeling than their European counterparts.

The relationship between regulators and polluters is by necessity delicate, and it is vital not to make it overtly antagonistic. Many companies find that the transaction cost of simply communicating and complying with the environmental agency's bureaucracy is a heavier burden (e.g., the U.S. Project XL, see Chapter 8) than the abatement costs themselves, which tend to be perceived as legitimate and to carry side benefits for the company in its relations with consumers and others. Being service-oriented and nonbureaucratic is important for environmental protection agencies but difficult when asymmetric information problems must be overcome and the temptations of corruption and nepotism avoided. In addition, many agencies struggle with fundamental problems such as funding of staff and equipment.

Supplemental Reading

Amacher 1996, 1998
Dasgupta, Hammond, and Maskin
1979, 1980
Dietz and Fankhauser
2010
Hoel 1998
Kwerel 1977

Macho-Stadler and Pérez-Castillo 2001
(Chapters 3 and 4)
Pindyck 2007
Roberts and Spence 1979
Segerson and Wu 2006
Shaw and Woodward 2008
Thomas 1995

Notes

- 1 For instance, uncertainty in abatement costs may arise because of technical progress combined with heterogeneity between firms and asymmetric information. Furthermore, the errors in judgment of abatement costs may be not random but strategic, because information presumably comes from the company itself.
- 2 Some diseases might be related to genetic factors and are assumed to be private knowledge; this is a case of pure adverse selection. If the risk of disease is related to lifestyle that the individual not only knows about but also can influence (e.g., diet), then the case also entails moral hazard.
- 3 This activity could be monitored at some cost; in fact, some automatic mechanisms are already in place. Swedish can-deposit machines distinguish between Swedish and Danish beer cans so that the latter cannot be inserted to collect the Swedish refund. In the United States, the refund for the same can may differ by state.
- 4 Modeling transaction costs as proportional to taxes raised, Smith and Tomasi (1995) show that taxes are not always superior and that the optimal regulatory policy is generally a mixture of taxes and standards. For a case of limited information (individual polluters cannot be monitored but ambient concentration is observable), Xepapadeas (1991) proposes budget-balanced contracts in which each polluter faces a random risk of an ambient fine (where the proceeds are distributed among the other polluters). Trading between point and nonpoint sources may save costs but raises issues of monitoring; Letson, Crutchfield, and Malik (1993) assess compliance by verifying technology and conclude that the optimal trading ratio depends on the relative costs of enforcing point versus nonpoint reductions, and on the uncertainty associated with NPSP. Shortle, Horan, and Abler 1998 is a good summary of NPSP research issues.
- 5 Thomas Sterner is grateful to Joseph Stiglitz for enlightening discussion of some issues discussed in this section, particularly labeling.

Equilibrium Effects and Market Conditions

An economic instrument has several effects. Consider a tax on a polluting input, which will directly affect input mix and technology in production, a direct effect which is often the main intended one. The tax also will have indirect effects: it makes the final product more expensive, and thus reduces demand, which is referred to as the output or output substitution effect. Finally, the tax raises revenue for the treasury.

In this chapter, we focus on such indirect effects. First, we discuss situations in which the indirect effects can be expected to be more important than the direct effects. Next, we discuss the revenue and other effects that are associated with the so-called double dividend debate. Many of these effects depend on market conditions. The existence of monopoly or cartels will affect and may completely distort the optimality of, say, taxes. The last section is therefore dedicated to a discussion of market conditions.

Goal Fulfillment, Abatement, and Output Substitution

The emissions from an economic activity or sector typically depend on its pollution intensity or rate of emissions (ξ) and its level of activity or size (q). Symbolically, $e = \xi q$ with the appropriate summation over firms and pollutants. Typically, the focus of environmental policy is on the pollution intensity or rate of emissions, ξ . In addition to technological measures for abatement, emissions can be reduced by decreases in output, and one of the advantages of certain market-based policy instruments is that, if correctly designed, they will lead to changes in output composition that reduce emissions damages. This advantage applies to taxes but not to subsidies. It also applies to tradable permits, as long as they are not allocated in proportion to output.

Whether output effects are important or marginal appears to depend on the ease of abatement as well as the demand elasticity for the product. Related issues are the precision of instruments, the degree of goal fulfillment, and

monitoring costs. Two contrasting examples are the use of chlorine to bleach paper products and carbon emissions, which lead to global climate change. In the case of chlorine, pollution intensity is relatively easy to reduce, and monitoring is fairly easy. In addition, the cost share of bleaching in paper production is small, and the demand for paper is fairly inelastic. All these factors suggest that if society wants to abolish the use of chlorine, it can choose an appropriate policy and reduce the use of chlorine with essentially no impact on the total use of paper¹. In fact, Swedish authorities used “no instrument” to achieve this goal. They merely signaled a serious intent to reduce chlorine use by drafting tax laws, and the paper mills “voluntarily” stopped using chlorine as a bleaching agent. If it had not been possible to monitor the use of chlorine and the issue were a major environmental problem, then a product tax on paper might have been considered the closest proxy available. However, such a tax would have been a very blunt instrument to deal with chlorine because it uses only the output effect and has no effect on pollution intensity.

In contrast, with carbon emissions, both factors can and should be influenced: the rate of emissions of fossil carbon per unit of energy, and the total use of energy. Abatement is one option but not the only one. Nonfossil energy (at the necessary scale) probably will be more expensive, and it is efficient to include reductions in overall energy use among the options chosen. Chosen instruments must have the correct output effect, which means that subsidies and voluntary agreements are not sufficient instruments. Furthermore, if exact goal fulfillment is required (e.g., because of obligations in international treaties), then quantitative instruments are favored.

Importance of Output Compared with Abatement

The importance of abatement cost and demand elasticity to environmental policy design is illustrated in Figure 11-1. In Figure 11-1A, abatement is cheap and product demand is inelastic, so technological abatement options are the best solution. In Figure 11-1B, where technological abatement possibilities and “clean technology” do not exist, the only mechanism to reduce pollution is to reduce demand and production. The effectiveness of this option depends on the demand elasticity of the product.

In Figure 11-1C, both abatement and output reduction are important. The cost increase that results from new and somewhat cleaner production methods is large, and environmental taxes must be paid (or permits bought); thus, the increase in the marginal cost of production is considerable. Because demand is fairly elastic, the increase in price due to the tax will result in a large decrease in output from q_a to q_0 in addition to the reduction from q_m to q_a caused by the direct increase in production costs due to abatement. Algebraically, the output and abatement effects of an instrument such as tax or permits (T) can be compared as follows: T tends to reduce output $q(T)$

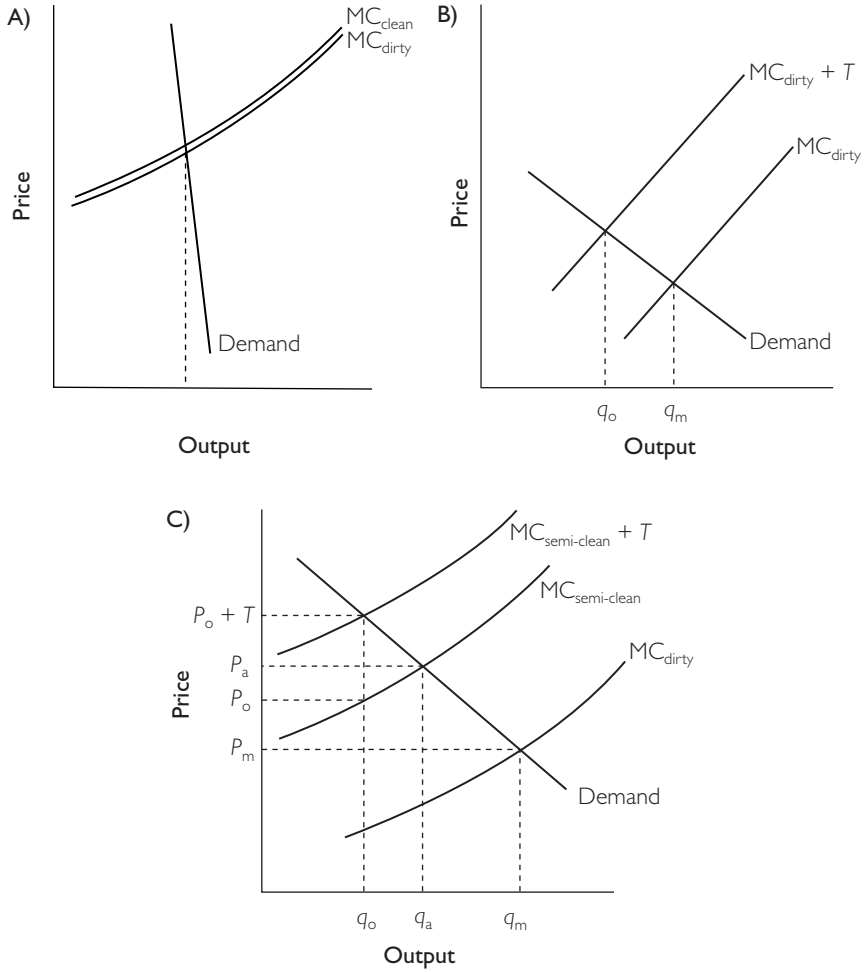


FIGURE 11-1 Abatement and Output Reduction

A: Inelastic demand, cheap abatement. B: Elastic demand, no clean technology. C: Elastic demand, clean technology. *Notes:* MC = marginal cost of production (for clean, semiclean, or dirty technology); P_0 = optimal production cost; T = tax; P_a = price if the optimal (semiclean) technology is used but no taxes levied; P_m = market price in the absence of any instrument; q_0 = product quantity (with q_0 , q_a , and q_m are defined analogously to P).

and rate of emissions $\xi(T)$, as in $e(T) = q(T)\xi(T)$. Differentiating gives $de/dT = qd\xi/dT + \xi dq/dT$; multiplying by T/e and substituting $e = q\xi$ gives the elasticity of emissions with respect to the instrument (ϵ_{e_T}) as the sum of the elasticities for output (ϵ_{q_T}) and abatement (ϵ_{ξ_T}):

$$\epsilon_{e_T} = \epsilon_{q_T} + \epsilon_{\xi_T} \quad (11-1)$$

The extent of the abatement response will depend on the sum of technological abatement and output substitution (i.e., the partial and general equilibrium effects). The output effect can be expressed in Equation 11-1 as an interaction of the demand slope, $q = q(P)$, and a shift in the supply curve, $P = P(T)$. Combining gives $q = q[P(T)]$ or

$$\epsilon_{q_T} = \epsilon_{q_P} \epsilon_{P_T} \quad (11-2)$$

and substituting into Equation 11-1 gives

$$\epsilon_{e_T} = \epsilon_{\xi_T} + \epsilon_{q_P} \epsilon_{P_T} \quad (11-3)$$

The total response in emissions depends on the possibilities of reducing the rate of emissions as well as the product of the price elasticity of demand for the final product and the tax elasticity of the marginal cost of supply (ϵ_{P_T}). As already mentioned, it depends on how much more expensive clean production methods are or on the abatement elasticity.

This relationship is nicely illustrated for the simple special case of linear additivity in Box 11-1. The tax elasticity of supply is equal to the share in total marginal costs of the marginal environmental tax (ϵ_{P_T}); these terms must be compared to the response in abatement technology or the rate of emissions, (ϵ_{ξ_T}). If the output effect is a large share of the total effect of the instrument, then there will be a big difference between using the tax and any other instrument that does not place the full cost of abatement and tax payment on

BOX 11-1 Tax Elasticity of Supply (Linear and Separable Abatement Costs)

Consider a firm maximizing profits equal to $Pq - C(q) - c(\xi_0 q - e) - T\xi$. Production costs are here additively separable and linear in abatement. Production (q) entails production costs $C(q)$ and a “normal” level of emissions ($\xi_0 q$) that can be abated to any desired level of emissions (e) by a linear cost function $c(\xi_0 q - e)$. The firm undertakes abatement to reduce environmental taxes ($T\xi$). Differentiating with respect to q and e gives first-order conditions $c = T$ and $P = C'_q + c\xi_0$ and, by substitution, $P = C'_q + T\xi_0$, where P is product price.

In this case, the increase in the supply price is the tax T times the original rate of emissions ξ_0 . Differentiating with respect to T gives $dP/dT = \xi_0$, and expressing this as an elasticity gives

$$\xi_{P_T} = \frac{T}{P} \frac{dP}{dT} = \frac{T\xi_0}{C'_q + T\xi_0} \quad (11-4)$$

Thus, in this simple linear case, the tax elasticity of supply is the relative share of the marginal environmental tax in marginal production costs. Equation 11-4 can be inserted into Equation 11-3, allowing for an exact determination of the relative size of abatement and output effects in this model.

the polluter. On the other hand, if the costs of abatement are small in relation to production costs, then the difference will not be as large. In the latter case, the use of other instruments or other ways of refunding taxes might be preferable. In small open economies or economies whose main goals are alleviating poverty and creating employment, the idea of addressing environmental issues through reduced output is bound to generate solid resistance (see Chapters 12–14).

Output Effects of Subsidies and Other Policy Instruments

The output effect—reduced emissions due to a higher product price—may be an essential part of the economic response to a policy instrument. (There may also be cases in which emissions increase, despite abatement reducing emissions per unit; but because output grows, this response is sometimes called the rebound effect.) The importance of the output effect was highlighted in an early analysis that focused on the perverse effects of abatement subsidies (Baumol and Oates 1988). In that model, the payments for emission reductions could exceed abatement costs and thereby lower total costs and increase output enough to counteract, and even reverse, the abatement effect. When marginal abatement costs are constant, the subsidy is simply a repayment of abatement costs. The marginal and average costs (calculated in Box 11-2) are

BOX 11-2 Marginal and Average Costs under Different Instruments

Assuming the same linear, additively separable cost functions as in Box 11-1 [total costs (TC) = $C(q) + c(\xi_0 q - e)$], the total costs for various instruments would be as follows:

without regulation: $TC_m = C(q)$

with taxes: $TC_T = C(q) + c(\xi_0 q - e) + Te$

with subsidies that exactly cover abatement costs: $TC_s = C(q) + c(\xi_0 q - e) + T(e - \xi_0 q) = C(q)$

with individual regulation (ξ_R): $TC_r = C(q) + c(\xi_0 - \xi_R)q$

Taking derivatives with respect to output (q), observing that $c = T$ gives marginal costs (MC) under taxes and subsidies, and dividing by q gives average costs (AC) as follows:

$$MC_m = MC_s = C_{q'}, \quad AC_m = AC_s = C(q)/q$$

$$MC_T = C_{q'} + c\xi_0, \quad AC_T = C(q)/q + c\xi_0$$

$$MC_r = C_{q'} + c(\xi_0 - \xi_R), \quad AC_r = C(q)/q + c(\xi_0 - \xi_R)$$

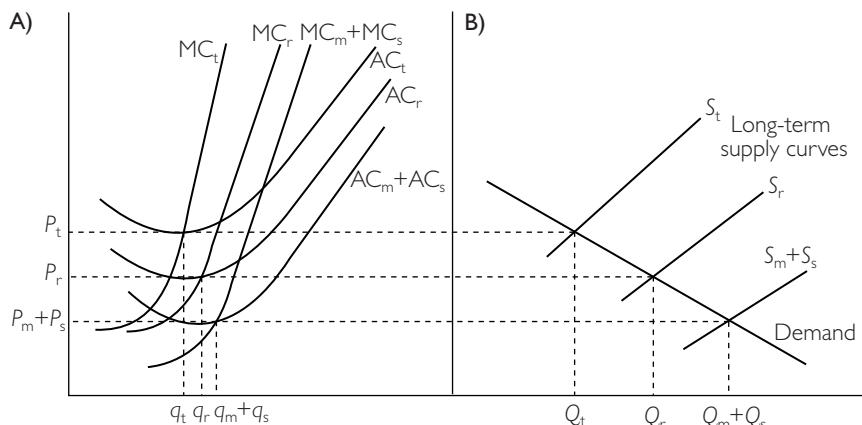


FIGURE 11-2 Effects of Subsidies and Taxes

A: Short-term costs for firms. B: Long-term market equilibrium for industry. Marginal abatement costs are constant, and the subsidy is equal to abatement costs. *Notes:* P = price; MC = marginal cost; AC = average cost; q = firm output; Q = total output; and S = supply or marginal cost curve. Subscripts s , t , r , and m indicate subsidies, taxes, regulated (or performance-standard) emissions rates, and “market” without any instrument, respectively.

illustrated in Figure 11-2. Because subsidies cover the abatement costs exactly, the output produced under subsidies is the same as (but not larger than) that in the unregulated case.

Figure 11-2A shows the short-run marginal costs (MC) and average costs (AC) for a representative firm without abatement as well as the corresponding curves with abatement induced by subsidies, charges, or performance-standard or regulated emission. With the subsidy construction chosen, the MC and AC curves are identical for a subsidy and the unregulated case². The tax causes the largest increase in MC and AC , whereas regulated abatement or emissions rates bring about intermediate MC and AC because the regulation affects abatement costs but not the costs for the unabated pollution. As illustrated in Figure 11-2A, the higher costs of taxes—and, to a smaller extent, the regulation—create a higher market price, which may or may not have an effect on firm output in the short term. (There is an effect, but it is small.) In the long run, the price is bound to reduce demand. In addition to firm output, the number of firms is endogenous, and the higher market price caused by abatement costs and taxes will lead to exit from the industry. A regulation of emission factors would not have the full output effect, even if the individual emissions rates were optimal. The subsidy instead will lead to a price reduction, delaying exit (and possibly encouraging entry)³ (see Lee 1999, Lahiri and Ono 2007, and Millimet et al. 2009).

General Equilibrium, Taxation, and the Double Dividend

In recent years, there has been a great deal of interest—perhaps excessive, in that it distracts from the task of managing the environment—in the so-called double dividend issue.⁴ Although taxes are optimal policy instruments in terms of allocative efficiency and incentive structure, they are often strongly resisted by those who have to pay. For this reason, “green tax reforms” were proposed, whereby the revenues from environmental taxation were to be used to reduce other (distorting) taxes. This idea can indeed be a good one, but some enthusiasts got carried away, suggesting that these proposals would thereby give double or even triple dividends of environmental as well as economic improvement (manifested as increased employment and growth).

To understand these issues properly, one needs a thorough knowledge of the theory of taxation. Only a few of the most basic principles of taxation are outlined here. If a particular consumption good is taxed, then its relative price will rise and consumption will decline. The consumer’s pattern of choices will thereby be distorted. This distortion gives rise to a loss in consumer surplus or excess burden of the tax, usually measured by the so-called Harberger triangles under the demand curve. Under certain conditions, the inverse elasticity rule states that only goods with inelastic demand should be taxed because the consumption of these goods is hardly affected and thus the distortion mentioned is small. (More precisely, it is the relative degree of substitution with leisure. Even if the demand curve is vertical, a product tax could reduce labor supply by increasing product prices and reducing real factor returns.)

The inverse elasticity rule applies literally only in simple models. The tax system has to take many factors into account; one of the more important is effects on income distribution. Because the demand for perfume, jewelry, and other luxury goods is fairly elastic, it may not be effective to tax these goods; however, they are consumed by rich individuals, and thus there may be equity reasons for taxing them. Taxation of luxury goods is particularly effective when it is not possible to tax wealth directly or to use nonlinear income taxes. In practice, it is fairly difficult to find goods with inelastic demand; one example is food (particularly among low-income groups), but taxing food more heavily than other goods is not acceptable from an ethical, distributional, or political viewpoint. In addition, differential taxation of goods entails many problems of practical administration, rent-seeking, and income distribution. Thus, the inverse elasticity rule is not always a very practical policy.

One special “good” is labor itself, and income tax is commonly used. In simplified models (without foreign sector, savings, and so forth), taxing income and taxing all consumption goods (a value-added tax) are equivalent. One might think that this approach avoids distortions, but it does not, because it may distort the allocation of time between labor and leisure. With high taxes, the incentives for (market) labor are diminished, and people may choose (excessive) levels of leisure (or nonmarket labor). There is a strong

contradiction between on the one hand wanting high and progressive taxation to fulfill goals related to income distribution, equity, the provision of public goods, and fighting poverty, and on the other hand the disincentive effects that such taxation may have on the high-income earners on the other.

A few additional “rules” of taxation are commonly discussed. One is to tax factors or goods that are not easily mobile (to reduce tax evasion); examples are land and forestry taxes and, to some extent, property taxes (although the size of housing may also respond in the long run). Another concerns the treatment of intermediate goods. It is generally not desirable to tax intermediate goods because certain inputs are taxed twice. Consider a market for home heating, which can be satisfied by using oil or electricity. If both are taxed by a uniform tax on consumption and the oil used to produce electricity is also taxed, then the home owner who buys this electricity would be taxed twice, thus distorting the choice between the two sources. In practice, having different tax levels for goods that can be either consumed directly or used as an input can be difficult (see the discussion on diesel taxes in Chapter 17).

The simple double dividend argument has been shown wrong in the following sense: if the environmental benefit (i.e., the first dividend) is disregarded, then an optimal tax structure can hardly be improved by lowering one of the tax rates—say, the rate on wages—and levying a supplemental tax (i.e., beyond what would be motivated by externalities or other market failures) that will give the same additional revenue on an environmental bad (Bovenberg and Goulder 1996). First, it is an illusion that the environmental tax is not a tax on labor; it is indeed a tax on wages because people use their wages to pay for whatever good generates the environmental disturbance (such as gasoline), and so, disregarding the externality, it is an inefficient tax on labor. The inefficiency that arises from lost consumer surplus due to a nonoptimal distortion of the consumption basket makes the tax effectively bigger in welfare terms than the labor tax it was supposed to replace. Thus, there is a direct welfare loss rather than a gain to this tax swap. To compound this injury, the decrease in effective real salary creates secondary effects, often assumed to reduce labor supply⁵. Furthermore, the environmental benefit (as opposed to the loss in private consumer surplus) is a kind of public good, and thus, considering the cost of public funds, its provision would be lower in a “second-best” optimum than the “first-best” reasoning would suggest.

It is thus incorrect to say that an environmental tax would be good, even if there were no environmental problem. This response has sometimes been used as a last line of defense for environmental tax proposals in which it is scientifically difficult to exactly measure the value of the ecological or health benefits. Interactions between the environmental tax and the preexisting taxes may be counterintuitive. Thus, the advantage of the dividends from the environmental tax are not greater in economies with high preexisting levels of (marginal) labor tax; rather, the high degree of distortion results in greater costs to public funds and bigger deadweight losses.

This attack on the double dividend neglects, however, to address the environmental externality. If it is addressed through regulations, then even these environmental regulations will have tax interaction effects; if labor taxes are high, then environmental regulations distorting consumption choices will lead to a large welfare loss and a significant social cost through reductions in labor supply. Taking this effect into account changes the picture somewhat. The relevant comparison is not only between an environmental tax and a general tax, because a general tax leaves the environmental problem unresolved; instead, an environmental tax should be compared with a combination of general tax and environmental regulation. In this comparison, the environmental tax proceeds are a positive rather than a negative factor.

An optimal environmental tax operates through five effects (Goulder et al. 1999):

- the abatement effect, which is fairly self-evident;
- the input substitution effect, which refers to the substitution of “cleaner” inputs for “dirtier” ones in production;
- the output substitution effect (discussed earlier);
- a revenue-recycling effect, a positive effect of the environmental tax that confirms what many people mean by a double dividend (i.e., if there is a cost to public funds, then an instrument that just happens to collect such funds must have some advantage due to this fact); and
- a tax interaction effect, which is related to the loss of consumer surplus and real wages that are due to distortion of the preferred consumption (or input) basket and which reduces labor supply and tax incomes, leading to further losses.

Between the last two related effects, the tax interaction effect may dominate over the revenue-recycling effect, but regulation and other instruments cause tax interactions, too. For command-and-control regulations, the tax interaction effect is a welfare loss, although small, because the price of the polluting good is not raised as much as with taxes. Overall, command-and-control policies lead to higher primary efficiency costs and higher second-best costs than other kinds of policies because the output substitution effect is not fully used and the sum of revenue recycling and tax interaction costs turns out to be higher (Goulder et al. 1999).

The earmarking of funds also must be discussed in this context. If environmental taxes are earmarked for use in an environmental fund, then presumably part of the revenue-recycling effect is lost (unless these funds happen to be the best use of the tax proceeds in that particular situation). The negative tax interaction could, for related reasons, be worsened.

One practical goal of using environmental taxes is the desire for a stable tax base, sometimes referred to as a revenue goal. Clearly, there is a contradiction of goals here: when abatement is easy, the pollutant emissions may go to

zero and thus the tax base would be gone (see Figure 11-3B). If the prospect of revenues from such a tax had been used to lower other taxes, there would be a shortfall for the treasury. Because changing the tax system is disruptive and expensive in itself, it probably is not optimal to apply taxes (as opposed to charges) to environmental problems that are amenable to abatement. Administrative charges or simple information and persuasion (because abatement costs are assumed to be small) may be a more appropriate instrument.

On the other hand, when demand for the polluting good is inelastic, the tax base is potentially stable (but there is little reduction in pollution; see Figure 11-3A). In this case, abatement is difficult, and an environmental tax

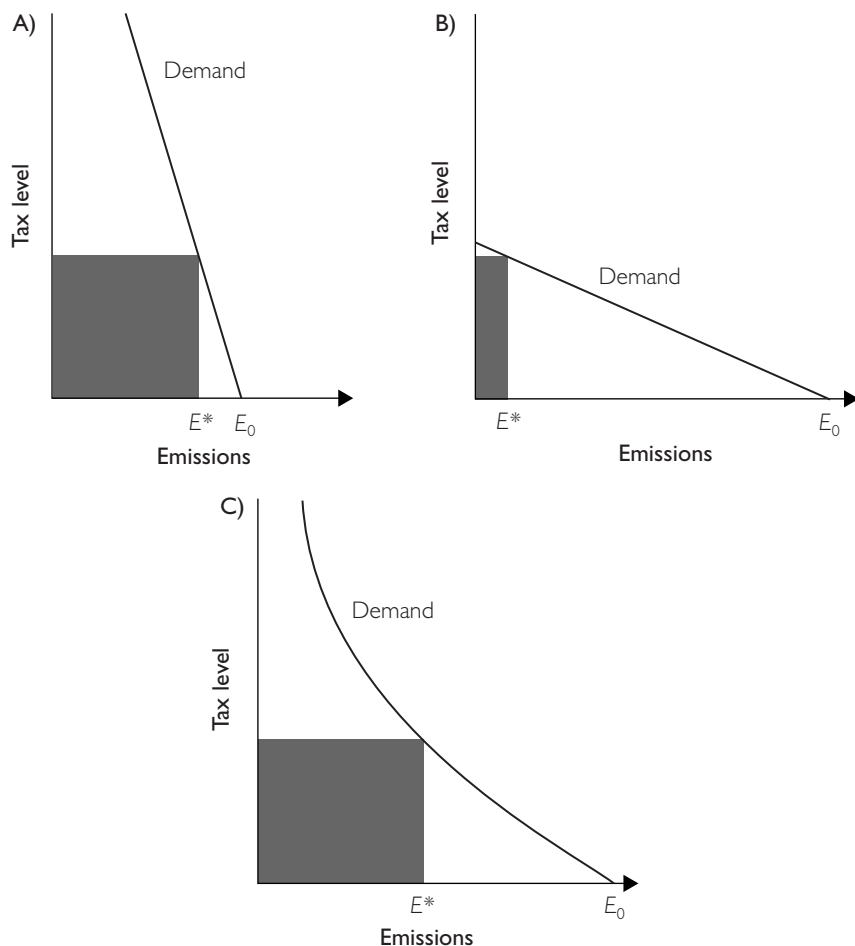


FIGURE 11-3 Environmental Taxes as a Tax Base

A: Steep demand, little effect on emissions, large tax base. *B:* Flat demand, large effect on emissions, small tax base. *C:* Nonlinear demand, large effect on emissions, large tax base.

Notes: E_0 = original emissions level, E^* = optimal emissions level.

might appear to be a failed policy. However, because the demand curve is steep, any other policy would also encounter difficulties in achieving significant abatement. At least the tax provides a continuous incentive for innovators as well as the output substitution and revenue-recycling effects. Presumably, some pollutants fall into an intermediate category (see Figure 11-3C), where the tax instrument performs well: first, it provides incentives for abatement through demand reduction; then (as demand becomes more inelastic), it provides a stable tax base, the revenues of which can be recycled or used for other purposes.

Adapting to Market Conditions

Up to this point, the discussion of policy instruments has assumed perfect competition, but such a description of the market structure is hardly exact in many markets—particularly not in developing countries. The problems of policy design under imperfect competition were reported more than three decades ago, when it was shown that the optimal charge for an externality produced by a monopoly would be lower than the standard Pigovian level.

The reason for this effect on optimal taxes is the existence of two imperfections: the monopoly (or oligopoly or cartel) that tends to indicate that optimal output would be higher than the output observed, and the negative externalities that pull in the opposite direction (see Figure 11-4)⁶. The

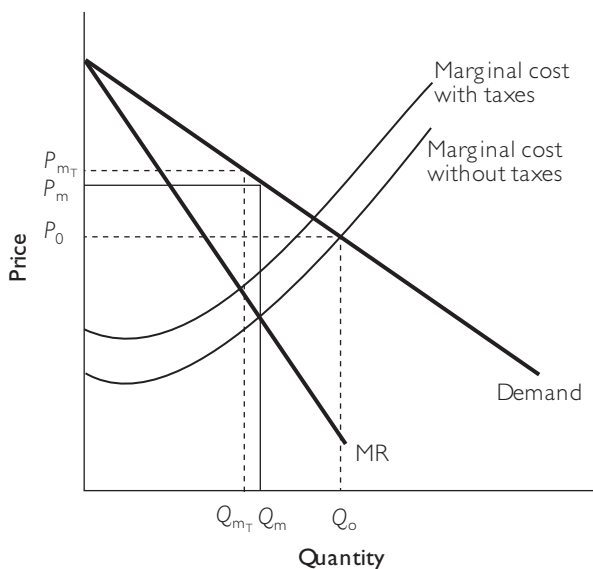


FIGURE 11-4 Environmental Taxes and Monopoly

Notes: P_m = monopoly price; P_{mT} = monopoly price if a Pigovian tax is (wrongly) applied; MR = marginal revenue; Q_{mT} = quantity sold under monopoly with tax; Q_m = quantity sold under monopoly; Q_0 = optimal quantity.

monopoly produces output Q_m at the monopoly price (P_m), and the social optimum would be a production of Q_0 , at a price P_0 . If a tax is set according to the Pigovian rule (equal to the sum of marginal damages), then the monopoly would further reduce output (that is already too low) down to the level Q_{mT} instead of increasing it. The social cost of the change in quantity is measured as the rectangle $[Q_{mT} - Q_m]P_0$. The consumer surplus lost is measured by the area under the demand curve $Q_{mT}P_{mT}P_mQ_m$. Since the latter area clearly exceeds the former, welfare has been reduced by the tax.

A second-best charge (calculated in Box 11-3) is $T_m = D'(e^*) - (P/|\xi|)Q'_e$, where $|\xi|$ is the absolute value of the elasticity of demand) but is not always feasible or even desirable. In the case illustrated in Figure 11-4, the second-best tax would be strongly negative. One might also directly discuss a combination of policy instruments: a subsidy to increase supply (to overcome

BOX 11-3 A Second Best Pigovian Tax

Let us assume an industry generating an externality. The industry is organized as a monopoly, with a single profit maximizing firm. Pigovian analysis suggests the levy of a corrective tax on T_m per unit of output. How should the regulator set the Pigovian tax in this second best setting? Maximization of welfare requires maximizing the difference between the social value of the monopolist's output and the full social cost. Let $P(Q)$ denote the inverse demand curve, $C(Q, a)$ is the monopolist's cost of production and abatement (where a denotes the level of abatement activity) and $D(e)$ denotes the social damages associated with waste emissions e . Thus, the regulator maximizes:

$$W = \int_0^Q P(Q)dQ - c(Q, a) - D(e) \quad (11-5)$$

Differentiating the welfare function with respect to the unit tax on emissions T_m , yields the following FOC:

$$P(Q) \frac{dQ}{dT_m} = \frac{\partial c(Q, a)}{\partial Q} \frac{\partial Q}{\partial T_m} + \frac{\partial c}{\partial a} \frac{\partial a}{\partial T_m} + \frac{dD}{de} \left[\frac{\partial e}{\partial Q} \frac{\partial Q}{\partial T_m} + \frac{\partial e}{\partial a} \frac{\partial a}{\partial T_m} \right] \quad (11-6)$$

The monopolist's problem is to maximize profits:

$$\pi = P(Q)Q - c(Q, a) - T_m e(Q, a) \quad (11-7)$$

Assuming that the monopolist takes the tax T_m as given, profit maximization implies the following FOC:

$$\begin{aligned} P(Q) &= \frac{\partial c(Q, a)}{\partial Q} + T_m \frac{\partial e}{\partial Q} - Q \frac{\partial P(Q)}{\partial Q} \\ \frac{\partial c(Q, a)}{\partial a} &= -T_m \frac{\partial e}{\partial a} \end{aligned} \quad (11-8)$$

Taking into account the monopolist's choice of output and abatement, the optimal tax becomes:

$$T_m^* = \frac{dD(e)}{de} + \frac{Q \frac{\partial P(Q)}{\partial Q} \frac{\partial Q}{\partial T_m}}{\frac{\partial Q}{\partial T_m} + \frac{\partial Q}{\partial a} \frac{\partial a}{\partial T_m}} = \frac{dD(e)}{ds} + \frac{-\frac{P(Q)}{|\varepsilon|} \frac{\partial Q}{\partial T_m}}{\frac{\partial Q}{\partial T_m} + \frac{\partial Q}{\partial a} \frac{\partial a}{\partial T_m}} \quad (11-9)$$

where ε denotes the price elasticity of demand. Notice that the first term on the RHS is the marginal social damages associated with an additional unit of emissions. This is equal to the standard Pigovian fee on a perfectly competitive setting. The second term reflects the welfare losses associated with the reduced output of the monopolist.

Note that the denominator can be interpreted as $\frac{de}{dT_m}$. Then, we can re-write the optimal tax as:

$$T_m^* = \frac{dD(e)}{de} - \frac{P(Q)}{|\varepsilon|} \frac{dQ}{de} = D'(e) - (P/|\varepsilon|)Q'_e \quad (11-10)$$

Thus, the second-best tax for a polluting monopolist should be less than for a firm in perfect competition. The tax will vary directly with the monopolist's price elasticity of demand. The more price elastic is demand, the smaller is the welfare loss associated with any reductions in output and the closer is the Pigovian to the marginal damage of emissions. There is nothing in this calculation that stops the "optimal" tax from being negative.

the "monopoly" effect) and a tax on emissions (to overcome the externality problem). In general, this approach will not be practicable, either. Subsidizing monopolists is hardly attractive. The way the problem is typically stated, environmental policymakers are about to implement charges when they realize that the polluter is a monopolist. This raises the question of why policymakers would not address the existence of the monopoly itself (e.g., by deregulation). When monopolies cannot be avoided (e.g., in some sectors of small economies), trade liberalization is the only viable alternative. However, it may be difficult in the presence of barriers to trade or high transport costs. In many countries, signs indicate that the classical monopolies are declining in importance as a result of globalization⁷.

When monopolies cannot be deregulated or broken up, regulators may have to choose between the inefficiency caused by a charge on a monopolist and the inefficiency caused by the externality, especially when many different industries produce the same environmental externality (e.g., carbon or sulfur emissions). If the tax works well in the competitive industries, it seems unlikely that policymakers would want to reduce the tax because of possible perverse effects in one oligopolistic industry. The relative importance of these two inefficiencies depends on the relative importance of abatement effects and

output substitution effects. With inelastic demand and a relatively low cost of clean technology (e.g., as in Figure 11-1A), a tax instrument might be satisfactory; a monopoly would involve too low a production level. An environmental tax would in this case have only a negligible effect in further distorting output. However, the tax would give incentives for cleaner technology. Finally, policy packages with combined instruments can be used. Refunded emission payments reduce the marginal net tax burden encouraging firms to increase output. If the marginal damage from pollution exceeds the marginal distortion in an imperfectly competitive output, then a refunded emissions payment (REP) leads firms to achieve the social optimum (Gersbach and Requate 2004). In a context with imperfect information and the possibilities of both moral hazard and adverse selection, the optimal policy might be a combination of an emissions tax, a subsidy per unit of production, and a lump-sum tax on profits (Laffont 1994a, 1994b).

Most industries do not belong to either of the ideal types (monopoly or “perfect competition”); they operate in intermediate markets with some degree of imperfect competition. In duopolies and other market structures, it is natural to frame the analysis in a game-theoretical framework. Results might be expected to be midway between the monopoly and competition cases, but numerous complex cases are possible. Complications arise, especially if industries make separate choices about technology and operation (e.g., Simpson 1995; Carlsson 1999, 2000; Katsoulacos and Xepapadeas 1995; Ebert and von dem Hagen 1998).

Furthermore, additional aspects must be considered, such as those related to the distribution of the burdens, transaction costs, political economy of decisionmaking, and power of the polluters. One elementary observation is that designing complex instruments such as tax laws, which must be approved by a legislative body, may be costly if regulators only want to regulate one firm (or a few firms). In such cases, direct negotiations are perhaps more appropriate. This approach appears to have been confirmed by the phaseout of leaded gasoline in El Salvador, a small country with only one refinery (see Chapter 18).

Another important consideration when choosing a policy instrument is to avoid increasing market power when there is limited competition in an input market. In such a setting, instruments may be used (or abused) by companies to build collusive agreements or impede the entry of newcomers. Several instruments convey such risks: subsidies for abatement, voluntary agreements, standards (particularly if the standards for old and new plants differ considerably), and the creation of property rights. Hahn (1984) analyzes the issue of market power in the context of markets for transferable property rights. His principal result is that the degree of inefficiency observed in the market is systematically related to the distribution of permits. For the case of one firm with market power, the results have some intuitive appeal. If a firm with market power would elect to buy permits in a competitive market, then it follows a strategy resembling that of a monopolist. If it would choose to sell permits, it follows a strategy resembling that of a monopolist. Trade will only

lead to an efficient outcome if the large firm's initial endowment corresponds to its efficient emission level. In this case, the large firm does not participate in trade and cannot distort the market price for permits. This does not mean, however, that large firms should not participate in trade. It rather means that the final allocation might be less efficient than the initial allocation, and whenever is legally and informationally feasible the regulator should allocate approximately as many permits to the large firms as they will ultimately need. In a recent paper, Wirl (2009) explores the interactions between noncompetitive sellers and noncompetitive buyers in a market for emission permits (i.e. oligopoly facing an oligopsony). In this setting, quantities demanded by buyers and offered by sellers are below the efficient levels. The double marginalization reduces trade from both sides compared with the competitive outcome.

Given the lack of empirical data, laboratory experiments have become an attractive tool to test for market power in emission trading. Studies have attempted to identify the welfare losses associated with two types of price manipulation: simple price manipulation—that is, manipulation in the permit price by firms that are competitive in the product market—and exclusionary manipulation, exercised by firms that hold market power in both the permits and product markets. Indeed, if the large firm is a monopolist in the permits market, it has the incentive to sell few permits: that not only keeps the price for permits high and earns high revenues when selling permits, it also increases the production costs of its competitors and thus reduces supply. On the other hand, if the firm is a monopsonist it has two opposing incentives: to buy few permits in order to hold down their market price, or to raise the competitors' costs by buying more permits than would be efficient. In a series of experimental treatments, Godby (2002) found that under simple price manipulation the efficiency gain through trade is approximately 60%–70% of the gains that are theoretically achievable through perfect competition; meanwhile, by contrast, under exclusionary manipulation permit trading might result in negative efficiency. In other words, permit trading turns out to be inferior to command-and-control policies. Nevertheless, it is worth pointing out that the subject pools participating in these experiments were mainly students and not real policymakers.

Finally, because recent years have seen an increasing emergence of green consumers, a question that arises is whether the performance of environmental regulations changes when consumers care more about the environmental impacts of the products they buy. Bansal (2008) analyzes the welfare implications of various government policies in a situation where consumers are environmentally discerning.

Two sources of distortion make the market solution differ from its social optimum: the oligopolistic power of the firms, and the environmental externality. The first leads to a larger quality gap among firms, which can relax price competition through product differentiation. The second leads to less than optimal quality choices. The welfare effects of any policy instrument depend

on the interplay between these two distortions. An ad valorem tax narrows the gap between the two qualities; it has a negative effect when increasing negative externality, but it has also a positive effect when it increases price competition and reduces prices. Bansal shows that the optimal policy instrument depends on the magnitude of the damage parameter associated with the environmental externality. When the externality is absent or the damage caused by it is sufficiently low, a small ad valorem tax is welfare improving. Instead, if the damage increases, the optimal policy shifts from an ad valorem tax to an ad valorem subsidy (see also Bansal and Shubhashis 2003 and Conrad 2005).

Supplemental Reading

Bansal 2008	Millimet et al. 2009
Bovenberg and Goulder 1996	Parry and Robertson 1999
Carlsson 2000	Repetto et al. 1992
Greenstone 2002	Xepapadeas 1997
Lee 1999	Wirl 2009
Mayeres and Proost 2001	

Notes

- Whether other bleaching agents (such as ozone) also cause problems is beyond the scope of this discussion. In principle, the prices of and markets for paper would be affected somewhat if the new processes were more expensive than the chlorine process, but this effect would be very small.
- Because of the definition of subsidy chosen and the constant marginal cost of abatement, the implications here are not the same as in the main case presented in Box 7-1 (see Chapter 7) or in Baumol and Oates 1988.
- Any subsidy will make average prices lower than in the case with taxes. The generous subsidy analyzed in Baumol and Oates 1988 makes them even lower than in the unregulated case and may lead to a perverse effect: that total pollution increases due to the subsidy. The Pigovian charge provides the correct incentive for the socially optimal number of firms in the long run (Spulber 1985). In a simplified form, for n identical and optimal firms, welfare can be written as $W = nPq - nc(q,a) - D[ne(q,a)] - nC_F$ where C_F is the fixed cost of entry. Other instruments can be analyzed analogously. Output-based allocation of permits or refunded emission payments will lead to results similar to that of regulated emissions rates: the marginal cost will be close to MCT, and the average cost is close to ACr. The long-run price will be close to P_r , reflecting the loss of some of the output effect of a tax.
- Many thanks to Ian Parry for comments on this section.
- If reducing the externality itself (e.g., congestion when driving to work) is complementary to labor supply, then the labor supply effects may actually be positive rather than negative (Parry and Bento 2001).

- 6 In a wide range of natural resource extraction models, this intuitive rule fails to apply (Brown 2000). In determining the optimal steady-state stock, prices do not enter the equation. The stock is decided by the intrinsic growth rate of the stock and the discount rate alone; market structure has no effect on optimal stock.
- 7 On the other hand, new firms with considerable market power are emerging, but they do not necessarily make their profits by reducing output and keeping prices up—as shown by Microsoft. Also, environmental policy instruments may be designed to speed up the process of deregulation. By taxing emissions rather than products, import competition would normally be strengthened.

Distribution of Costs

Policy instruments not only cause different allocative effects but also impose different financial burdens for polluters, victims, and society. The distribution of costs is distinct and thus linked to concepts of rights (see Chapters 3 and 8) and to the politics of policymaking. In this chapter, we focus on two kinds of cost distribution: first, among polluters, victims, and society, and then among the polluters themselves before we address the effects on income distribution in general.

First, consider the victims of pollution. The Coasian perspective emphasizes that parties should be left to negotiate with minimum support from the state (see Chapter 3). The size of the damages depends on many variables, including the number of victims; in a sense, a polluter may become “swamped” with “victims” who choose to locate near the source of pollution, which commonly is also a source of employment. A company may have built a plant with small externalities as measured conventionally, but when people move in, the external effects become larger. If environmental charges are based on the total value of such externalities, then it can be argued that the victims impose an externality on the polluter. However, in practice, negotiations are not effective with large numbers of victims. One exception is the health-related claims brought against U.S. tobacco firms, but the number of agents was not as large as the number of smokers, because the states sued. In this kind of setting it may be reasonable to assume that victims are represented by society (see Chapter 15), providing a rationale for general policymaking on behalf of the victims.

Several factors influence regulators’ choice of policy instrument. Typically, the focus is mostly on efficiency, and insufficient thought is given to distributional and political economy aspects. Different instruments can lead to significant differences in the distribution of costs and thereby in the support or opposition that policy proposals engender. Efficiency addresses the total cost of abatement, which is the relevant variable for aggregate welfare analysis, but small differences in the distribution of the costs may often be politically more sensitive than fairly large differences in total cost. The managers of large firms today are often primarily judged by and concerned with their relative

performance vis-à-vis competitors or vis-à-vis the market average. This relative performance is an important measure of their success. The typical polluter might thus be concerned with two distinct aspects of the distribution of the costs: its impact on the whole industry and (generally more important) its relative impact, that is, compared with competitors. With market-based instruments (MBIs), firms with the lowest marginal abatement costs typically do more abatement and thus end up having to bear higher total abatement costs. Because the level of abatement is chosen voluntarily by the firm, these costs must be more than compensated by the side payments (the difference in taxes or purchase of permits) involved in an MBI.

Distribution of Costs and Rights between Polluters and Society

The distribution of costs and benefits of instruments between the aggregate of polluters and the aggregate of victims and society can be discussed with the help of a simple illustration (see Figure 12-1). The regulator wants to reduce aggregate emissions from an initial level E_0 to a socially optimal level E^* , where marginal abatement cost (MC_a) equals the marginal cost of damage to the environment (MC_e). The benefit of environmental improvement from the emissions reduction is equal to the sum of areas D (which also represents abatement costs), F, and G. The net gain to society is thus always $F + G$, but the distribution of costs between parties varies with different instruments. Area C is the cost of unabated pollution; $B + C$ is the classical Pigovian tax or scarcity rent, and B is thus a financial transfer from polluters to society when using taxes

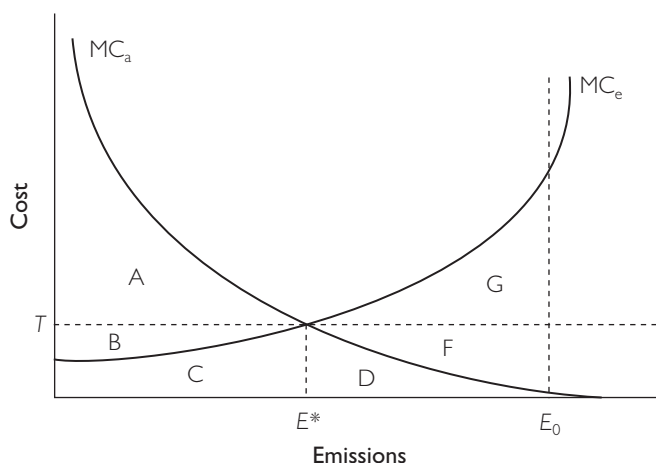


FIGURE 12-1 Environmental and Abatement Costs Associated with a Reduction in Emissions

Notes: MC_a = marginal cost of abatement; MC_e = marginal cost of damage due to emissions; E^* = optimal emissions level; E_0 = original emissions level; T = Pigovian tax level; A–G = areas that represent different cost components discussed in the text.

or auctioned tradable emissions permits (TEPs) as instruments. Similarly, D + F would be the size of a subsidy if the state paid the polluter a fixed subsidy T for each unit of abatement. Only area D corresponds to costs, whereas area F is a financial transfer from society to polluters if a fixed subsidy per unit of abatement is paid. However, subsidies are often designed to cover only D or part of D. The transfers B and F are a consequence of using a single market price on the environment even though the marginal costs are not constant. Figure 12-1 can also be interpreted to cover users of ecosystem resources such as water or air or even jointly owned resources such as a fish stock.

Depending on the choice of instrument, the various costs illustrated in Figure 12-1 are distributed differently between the polluters and the victims of environmental damage. The top part of Table 12-1 shows the different distributions of costs and benefits that achieve the same abatement, that is, bringing the level of emissions from the initial E_0 to the optimal E^* . The environmental benefit is the same for all columns. The costs (or payment) borne by polluters and society include physical abatement costs as well as tax or subsidy payments. For the sake of simplicity, society represents the victims of pollution. (In a fuller treatment, we could distinguish between situations in which some victims of pollution are compensated and some are not.) The bottom half of Table 12-1 presents the instruments best suited for meeting each definition of environmental ownership.

TABLE 12-1 Distribution of Costs with Different Instruments and Environmental Rights

Instrument	Ownership rights to the environment				
	Polluter (absolute)		Polluter (relative)	Mixed	Victim (polluter pays principle)
	(1)	(2)	(3)	(4)	(5)
<i>Distribution of costs and benefits</i>					
Environmental benefit	D + F + G	D + F + G	D + F + G	D + F + G	D + F + G
Polluter costs	F	0	D	(C + D)	(B + C + D)
Costs to society	(D + F)	D	0	C	B + C
<i>Type of instrument</i>					
Quantity-type	Public cleanup		CAC, VA, TEP (free)	TEP, partly auctioned	TEP, auctioned
Mixed			Hybrids	Hybrids (e.g., TEP + tax)	Hybrids
Price-type	Subsidies		REP, Tax subsidy	Partly REP	Tax, DRS

Notes: CAC = command-and-control policy; VA = voluntary agreement; TEPs = tradable emissions permits; REPs = refundable emissions permits; DRS = deposit–refund scheme. Single letters refer to areas in Figure 12-1.

Whether the polluter or society should bear the costs of pollution is a question with aspects of efficiency, welfare, and ethics. In the spirit of Coase (1960), the question can be approached as a problem of how to define property rights to the environment. The columns in Table 12-1 refer to different ways of distributing the cost burden between the polluter and the victim (i.e., corresponding to an equivalent set of concepts of ownership rights). When the polluter has exclusive and absolute ownership rights to the environment, a victim (or society) who wants a clean environment will have to pay for it—either through abatement subsidies (if a price-type instrument is chosen) or through a publicly financed cleanup (if a quantity-type instrument is chosen) such as municipal trash pickup. In Column 1, subsidies are paid for every unit of pollution abated. Thus, society pays the “extra” subsidy (F) to the polluter, who in this case does not pay anything but earns money. In Column 2, abatement subsidies only cover actual abatement costs (D ; in practice, it may only be a part of D , such as when credits for abatement investments are partially subsidized by low interest rates).

Column 5 of Table 12-1 represents the most far-reaching interpretation of the “polluter pays” principle. The ownership right resides squarely with society (perhaps in representation of the victims of pollution), and polluters will have to pay the equivalent of a market price for the right to use “environmental services.” The price-type instrument to reflect this type of ownership would be a conventional (Pigovian) tax, and the quantity-type instrument would be an auctioned TEP scheme.

In Column 3, the polluter has limited, intermediate rights—the polluter may use the environment or resource in question freely but has to maintain it in some acceptable condition. The polluter must pay for some reasonable level of abatement (D) but not for the pollution caused at that level. This distribution of costs may be law-based (the polluter or resource user has the rights), or it may be a recognition of the relative clout of the different parties. Either way, the regulators have the option of choosing ordinary regulation (design or performance standards), voluntary agreements (VAs), a TEP scheme in which the permits are allocated free of charge, a tax–subsidy scheme, or a refunded emissions payment (REP) scheme. Intermediate instruments (e.g., TEPs with a refundable payment for excessive pollution) also are conceivable.

Column 4 represents an intermediate concept of ownership and shares the cost burden between the polluter and the victim. In this interpretation of rights, polluters must pay the costs of abatement and unabated pollution but not the “extra” financial transfer (area B) associated with a single market price on environmental services. It could be approximated by a TEP scheme in which permits are grandfathered up to a certain total emissions level and auctioned thereafter; or, permits could be grandfathered but carry a charge or tax. This approach has been used in the United States for allocating permits for chlorofluorocarbons and halons, which were taxed by the U.S. Congress when it became apparent that this program conferred large monetary benefits

to the permit recipients (Tietenberg 1995; see also Chapter 20). Similarly, emissions payments could be partially refunded and, for example, a share of the revenues set aside to pay the costs of unabated pollution. This scheme is similar to some of the fee systems used in formerly planned economies and in developing countries, where fees are paid only in excess of certain levels and then paid into various environmental funds with different (regional or other) scopes and mandates (see Chapter 21).

A TEP scheme could allocate permits to current polluters (in proportion to output) or to historic polluters. The output allocation corresponds more closely to the property rights implied by the REP. One interpretation of rights is that they ultimately belong to society but that the polluter earns these rights by producing something useful or desirable for society. The grandfathered rights correspond better to a classical “prior appropriation” concept of rights because they are given to the polluters who had them in the past. For polluters, REPs are preferable to a tax or charge. Similarly, the exact allocation mechanism for TEPs (auctioned, grandfathered, or output allocated) as well as the finer details of how the mechanisms are implemented—such as the choice of baselines and starting years—make a big difference to the firms involved, and these aspects tend to dominate practical attention.

Instrument choice is commonly discussed as if the main choice were between grandfathered permits and taxes. These mechanisms lead to different consequences for the distribution of payments. Thus, the distributional issue can get mixed up with the choice between price- and quantity-type instruments. The point of Table 12-1 is to show that regardless of who has ownership rights and who should bear a certain share of costs, a broad choice between price- and quantity-type instruments still remains. Other instruments also are related to the material presented in Table 12-1. Deposit–refund systems belong with taxes in Column 5. VAs are listed in Column 3 but also can be applicable in Column 2, because the firm only has to pay partial costs of abatement. Liability is more complex because it introduces the probability of damage (and detection). However, strict liability appears closest to the concept of rights in Column 5, whereas negligence liability would correspond to the more intermediate concept of rights in Columns 3 and 4. Informational instruments and labeling do not have such a clear position but are presumably most applicable with concepts of intermediate rights. They definitely do not fit (as the sole policy instrument) with the polluter pays principle in Column 5.

Allocation of Rights

When a permit system is used, decisionmakers must decide how to allocate the permits. For REPs, the corresponding problem is how to refund the charges. Because the permits can have considerable value, a great deal of interest is focused on their allocation. In the 1999 allocation process for summertime

U.S. nitrogen oxide permits in 22 eastern states, the total estimated value of the permits was US\$1 billion to US\$2 billion. The efficiency gains that tend to be the main focus of economics may be larger in absolute magnitude, but their incidence is spread across many producers and consumers over a long period of time, and they are thus to a large extent “invisible.” However, the permit values are to be distributed among a small number of firms and are highly visible. Thus, they inevitably tend to attract most of the attention.

Pollution “rights” (explicit for permits, implicit for refunds) can be allocated in several ways. First, it must be decided how large these rights are: do they cover the entire environmental “rent” ($B + C$ in Figure 12-1), part thereof, or even more (corresponding to areas D and F)? Auctioned permits or taxes build on the idea that society, as represented by government, originally owns the rights. This method has several positive properties but tends to be resisted by polluters—especially in cases of congestion, in which the polluters or resource users (e.g., motorists, fishermen) are (almost) identical to the victims. Even though congestion can be solved in an allocatively optimal manner through auctioned permits, the agents are worse off with a tax or auctioned permits than with no policy; more rent is available to society, but it is all appropriated by the government, leaving nothing for those agents it was supposed to benefit.

Two basic choices must be made in designing the allocation principle: whether to use historic or current variables, and whether to base the allocation of rights (or refunds) on output, emissions, or some other variable. To enhance long-run efficiency (with respect to output and entry effects), polluting firms must include the opportunity cost of pollution rights in their calculations. However the rights are allocated, the allocation must be perceived as definitive. Renegotiation with respect to current variables should not be possible. The independence of current variables guarantees the absence of perverse effects on business decisions; it is one reason for the success of cap-and-trade permit programs, even though it reduces the flexibility of the system from the viewpoint of politicians and environmental protection agencies.

The historic variable most commonly used as a basis for allocation of permits is levels of emissions or, for natural resources policy, harvest levels, which ties in well with the legal doctrine of prior appropriation. Examples include the grandfathering of free permits and the tax–subsidy scheme (proposed in Pezzey 1992 and discussed in Chapter 7). The latter is instructive in this connection because the polluters have to pay a tax on emissions, except that they are given a certain baseline level (related to their historic levels) to which they have a right; they must pay the charge on emissions only in excess of this level. If their emissions are below the baseline, the instrument effectively becomes a subsidy.

This principle is naturally popular with polluters, but one disadvantage is that it does not encourage “proactive” behavior; as such, its use is undesirable. In fact, if all polluters knew that regulators would always grandfather

the allocation of permits, then they would never have an incentive to clean up problems in anticipation of future legislation and policy. On the contrary, polluters would have a strong incentive to emit as much as possible in order to get a large allocation of permits or a “high” baseline against which to measure future abatement. If all firms in an industry acted similarly, it would be difficult to find demonstrated clean technology, and the development of clean technology would be slow.

The undesirable feature of historic variables can be partly compensated for by choosing baselines many years prior to the decision, but this method has other practical disadvantages: measurements may be inaccurate, old values tend to be thought of as outdated, and newcomers will be highly critical. Newcomers tend to be the most progressive (in terms of efficiency and environment), and a policy instrument that supports the oldest and most polluting industries to the detriment of the newer ones is unsuitable, at least in that respect.

In the context of global warming, the concept of prior appropriation is fiercely contested by countries with large populations and small (but rapidly growing) levels of pollution, which would be more favored by a per capita or needs-based allocation (which might also be favored by countries with cold climates). Some have even emphasized the stock characteristics of the pollutants, arguing for a per capita allocation of total emissions (historic, current, and future), thereby leaving little allocation in the future for the industrialized nations.

When the carrying capacity of an ecosystem varies rapidly and unpredictably and is threatened by a market failure such as overuse, then policy instruments must meet two distinct goals: first, they must be efficient, which might mean that there is a strong need for permanent and verifiable property rights (permits or quotas); second, ecological factors require some degree of flexibility. A good example is found in the case of fisheries. Fishermen need quotas that are private property in order to plan their economy (e.g., take loans). At the same time, stock dynamics of fisheries are such that total allowable catch must be varied each year. This has been solved (as already mentioned) by denominating the individual transferable quotas in terms of shares in total harvest. The fishermen have fixed shares (which can be sold, inherited, or used as collateral), but the total catch can be set by a board of fisheries experts based on recent assessments of stock (see Chapter 25). If the world had to implement a full allocation of permits for carbon emissions into the atmosphere for the whole of this century, it would be wise to include some similar mechanism; if it were later determined that the amount of emissions to be reduced was more than allocated, then society would face an extremely costly burden of buying back the allocated permits.

To circumvent the somewhat arbitrary nature of an allocation based on historic variables, current variables can be used. Current emissions are not a good measure to use. However, two natural ways of allocating permits are 1) in proportion to a key input (such as fuel) or 2) to output, which is the same

as regulating emissions rates rather than total emissions (see Chapter 20 on the U.S. lead rights program or Chapter 20 on the Swedish policy of refunding charges on NO_x).¹ One disadvantage of these approaches is that they provide an effective “output subsidy” to the firms: if the refund or permit allocation depends on a variable that the firms can influence (such as output), then firms may be perversely encouraged to overproduce. (For monopolies with suboptimally low supply, an output subsidy might actually be more appropriate.) The ordinary command-and-control approach (e.g., best-available technology) has similar properties.

Incidence of Costs among Polluters

Another important area in which the distribution of costs may be important is among polluters. Although the sum and the distribution of polluters’ abatement costs should be the same with any MBI (at least to a first order of magnitude; “wealth” and entry and exit effects or effects of strategic behavior might be distinct), the distributions of transfer payments among the polluters are usually different depending on the permit distribution or refund mechanism chosen. The distribution among polluters is important not only for reasons of fairness but also because it affects political support for and resistance against certain kinds of instruments.

Grandfathered permit allocation favors firms that have been large polluters in the past and disfavors newcomers, which are likely to use more modern techniques and be more efficient. Taxes, auctioned permits, and REPs are relatively more favorable to firms with low emissions rates, and such firms may conceivably approve of these instruments despite their cost, because competitors are faced with even higher costs. In Figure 12-2, $a_2 WY\hat{a}$ is the net savings in abatement for Firm 2, which has higher abatement costs than Firm 1; $\hat{a}RXa_1$ is the increased abatement cost for Firm 1; and the difference between these costs is the net savings to society in reduced abatement costs.

To realize these cost savings, Firm 1 must be motivated to bear the extra abatement cost, and MBIs must be used: firms must receive subsidies, pay taxes, or trade permits. Whichever mechanism is used, the difference in payments between the two firms is $a_2 WXa_1$; if an environmental tax is used, the taxes are higher for Firm 2 by the same amount; and if permits are used, then the value of the permits is half of $a_2 WXa_1$.² The amount of the cost savings depends on the heterogeneity in abatement costs (see Chapter 9), and the level of the taxes depends on the amount of the emissions ($O'a_1$ and $O'a_2$). The relative sizes of these amounts can vary considerably. For some parameter values (low heterogeneity in abatement costs—see Box 12-1), the transfer payments caused by the MBIs between agents will be much larger than the net gain in reduced savings. In these cases, distributional issues will be thought of as more important than overall allocative efficiency.

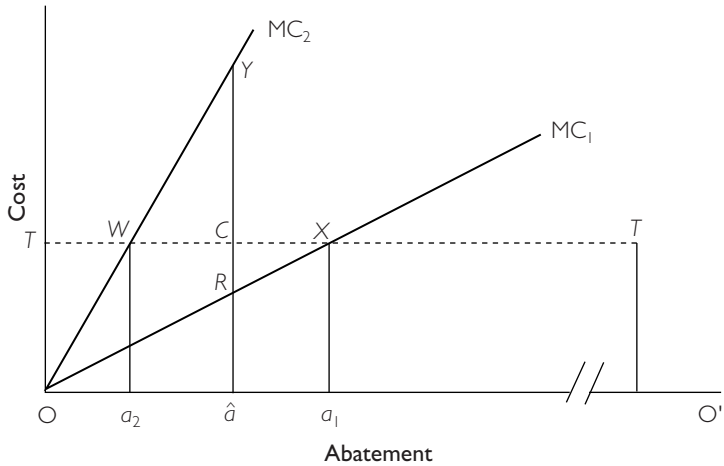


FIGURE 12-2 Cost Implications to Firms of Different Market-Based Instruments

Notes: MC = marginal cost of abatement for Firms 1 and 2; T = Pigovian tax level; a = optimal level of abatement for Firms 1 and 2; â = equal level of abatement for both firms; O = origin.

BOX 12-1 An Example of Abatement and Permit Costs for Two Firms with and without Trading of Permits

Taking the linear marginal cost functions ($MC_1 = a_1$ and $MC_2 = ha_2$) from Chapter 9 as an example, the costs for each firm can be calculated. With equal abatement reductions ($\hat{a}$), Firm 1 pays $\frac{1}{2} \times \hat{a}^2$ and Firm 2 pays $\frac{1}{2} \times h\hat{a}^2$ (see column 3 in the table below). Total abatement costs are $\frac{1}{2} \times \hat{a}^2(h + 1)$. If trade is allowed, then Firm 1 will abate $2h\hat{a}/(1 + h)$ and Firm 2 will abate $2\hat{a}/(1 + h)$ (see column 5). The MC of abatement is $2h\hat{a}/(1 + h)$, so total costs of abatement are $2h^2\hat{a}^2/(1 + h)^2$ and $2h\hat{a}^2/(1 + h)^2$, respectively (OW_{a2} and OX_{a1} in Figure 12-2). Total abatement cost savings are $\hat{a}^2(h - 1)^2/2(h + 1)$. Total costs for each firm consist of abatement and permit costs (columns 5 and 6). The total difference in cost to the two firms is $2h\hat{a}^2(h - 1)/(1 + h)^2$, which is bigger than the total resource savings in abatement costs even for quite large values of h (roughly, under 5).

Abatement and Permit Costs for Two Firms with and without Trading of Permits

Firm no. (1)	Equal reductions			Equalized MC after trading permits		
	Abatement (2)	MC (3)	Cost (4)	Abatement (5)	Abatement cost + permit cost (6)	Total cost (7)
1	$\hat{a}$	$\hat{a}$	$\hat{a}^2/2$	$2h\hat{a}/(1 + h)$	$2h^2\hat{a}^2/(1 + h)^2$ $2h\hat{a}^2(h - 1)/(1 + h)^2$	$2h\hat{a}^2/(1 + h)^2$
2	$\hat{a}$	$h\hat{a}$	$h\hat{a}^2/2$	$2\hat{a}/(1 + h)$	$2h\hat{a}^2/(1 + h)^2$ $2h\hat{a}^2(h - 1)/(1 + h)^2$	$2h\hat{a}^2/(1 + h)^2$

Notes: MC = marginal cost; â = abatement reductions; h = heterogeneity.

Additional complexities enter into the picture in empirical cases. For environmental taxes, there may be exemptions; for subsidies, baselines have to be formulated; for permits, an allocation mechanism must be formulated and implemented. The exact allocation of permits is often the result of a long and complicated political process and is difficult to foresee exactly in advance (Hahn 1990; Stavins 1998). Subsidies, output-allocated permits, and REPs entail some distributional consequences of the output effects discussed earlier. Grandfathering tends to benefit (the shareholders of) older, heavily polluting firms, whereas output allocation (or REPs) is more beneficial to cleaner newcomers.

Efficiency dictates that marginal costs be equalized, so firms with low marginal costs of abatement bear high total abatement costs. How does an environmental protection agency persuade one firm to bear higher costs than its competitors if not through side payments (as in the MBIs)? If marginal abatement costs differ because Firm 1 has historically done much less abatement, then Firm 1's higher requirements might be thought of as "reasonable." However, if that is not the case, and the difference in marginal abatement costs is due to other factors, then the firm probably will not voluntarily agree to undertake more abatement than Firm 2 without receiving some kind of compensation. Regulation would be much less costly to society if it were possible for an environmental protection agency to make informed, reasonable judgments and arrive at an approximately optimal allocation of abatement. This goal might be achieved with VAs, which are sometimes suggested as an alternative to MBIs, although the likelihood is very low.

Instruments that are sensitive to the distribution of the costs of abatement and transfer payments appear to be at least as relevant for developing and transitional economies as for developed economies (Chapters 3 and 4). The reasons for this relevance include higher emissions levels, the less competitive nature of the economy (which increases interest in strategic effects), and inadequately developed credit markets (which can make abatement difficult to finance).

In the context of climate change policy, several studies have analyzed the incidence of a price on carbon induced by a cap-and-trade program or carbon tax in United States. As expected, in the case of emission permits the distributional effects hinge crucially on whether permits are grandfathered or auctioned, and whether revenues from permit auctions are used to cut payroll taxes, to cut corporate taxes, or to provide lump-sum transfers. Parry (2004) finds that grandfathered permits are regressive since they redistribute income to wealthy households by creating firm rents that ultimately accrue to shareholders. Nevertheless, the regressivity is also explained by the fact that low-income groups have relatively high budget shares of polluting goods. Thus, carbon policy seems strongly regressive even in the case of carbon taxes or auctioned permits, where the recycled revenues are used to compensate bottom-income quintiles. In a similar vein, Grainger and Fullerton (2010) analyze how consumption differences across income groups drive the

regressivity of carbon policy. By using data on consumption, emission factors, and the structure of the U.S. economy, they calculate how a carbon tax is distributed across income groups. They assume that companies facing such a tax can pass on the compliance costs through changes in consumer prices, stock returns, wages, and other factors of production. In line with Parry's results, they find that a carbon tax is regressive; fuel consumption and electricity use are the main drivers of the regressivity of a carbon tax (see also Hassett et al. 2009).

Distributional Consequences of Congestion Pricing

The distribution of costs and benefits can sometimes make a welfare-enhancing policy quite difficult to implement. Consider Figure 12-3, which depicts the case of a congested road (or a town or any other congested resource—even a fishery for that matter). Private marginal costs are low (see the curve *abe*). Taking congestion and then other environmental costs into account gives successively higher marginal *social* costs. As can be seen readily from the curve, the unregulated “market” outcome at *e* is inferior to a regulated optimum at *h*. The benefit to society of the introduction of a fee equal to *bh* per vehicle

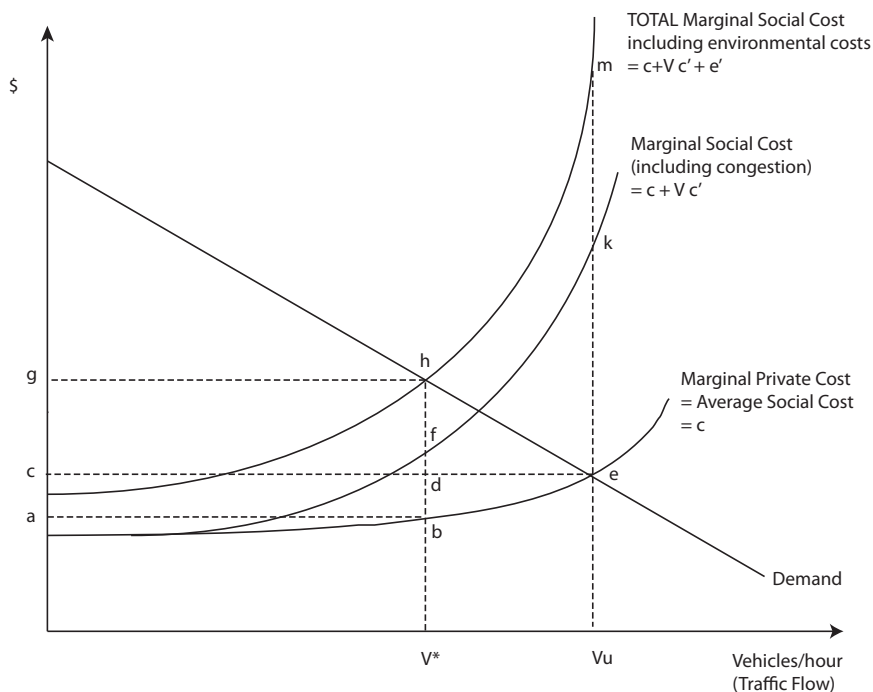


FIGURE 12-3 Distributional Aspects of Congestion Pricing

Note: Stylized model for demand and marginal (private and social) supply costs for vehicle miles travelled.

driven is the avoided welfare loss hem , but one should note the complicated set of distributional consequences:

- The victims of pollution gain $fk mh$
- The state gains tax revenue $ab hg$
- Motorists who continue driving gain time due to decreased congestion, but pay the tax. The welfare gain is represented by the rectangular $abdc$. The total tax payment is $ab hg$. Then, the net effect is a loss equal to $-cd hg$.
- Motorists who stop driving lose benefits equal to V^*V_{eh} , but enjoy a reduction of private costs of V^*V_{ueb} , making a net loss equal to beh .

Income Distributional Effects and Poverty

Three kinds of costs are involved in pollution: environmental damage, abatement costs, and expenses related to the instrument chosen (e.g., transfer payments, information, and administration). Each of these categories has effects on distribution. Many of the most important effects of policy instruments on income distribution are related to the cost factors discussed in the previous sections. The distribution of costs has effects on employment, rents, profits, and taxes.

Environmental programs are sometimes accused of putting firms out of business and creating unemployment for the sake of protecting national parks and other ecosystems that provide amenities and existence values to the already privileged. This claim may, in a few cases, be true; because privileged individuals tend to be more vociferous, their environmental projects may be more likely to be funded. In principle, some other projects might support ecosystems that poor individuals rely on or enjoy, and a particular allocation of costs might disproportionately affect rich individuals. Distributional issues with respect to individuals builds heavily on the effects already analyzed in the previous section, because most individuals are affected as workers, consumers, or owners of property or shares.

In general, little can be said about distributional impacts except that they must be analyzed on a case-by-case basis. The distributional impacts of environmental damages are completely separate from the corresponding impacts of abatement or instrument costs. Any of these components might be progressive, neutral, or regressive in general terms, and any kind of cost may or may not affect the welfare (in benefits or damages) of specific groups.

The provision or enhancement of environmental amenities that are luxury goods tends to have a regressive profile. Although it may apply to some cases (epitomized by scenic beauty or recreation), it is far from being a general truth. The poorest segments of society generally are the most dependent on intact common property resources for their livelihood and survival in terms of food, fuel, building materials, and medicines. A program that seeks to maintain mangrove swamps, for example, will benefit the poor individuals who

gather various products there. The same program may increase costs to companies producing shrimp in shrimp ponds, but those companies are likely to be wealthier than the individuals who are dependent on the resource. (However, shrimp farms may employ low-income people, so the issue is not clear.)

Similarly, it is the poor who are most affected by bad water quality because they cannot afford bottled water, filtering systems, inoculations, and other medical treatment. Projects that protect sources of drinking water thus have a progressive distribution profile.

Despite these counterexamples, many environmental projects may have regressively distributed benefits. In a similar way, abatement costs may be distributed progressively or regressively. Generally, the agents who produce the pollution are the ones who will be most affected by abatement costs. The choice of instrument also can make a big difference: an instrument such as an environmental tax or auctioned permits puts the burden explicitly on the polluter. However, instruments such as grandfathered and output-allocated permits, REPs, and subsidies have different cost distribution properties. The structure of a market determines whether a polluting company can pass on the costs of abatement and regulation to its consumers or back to its workers and suppliers earlier in the production chain. With information on the distribution of costs and benefits, this information must be considered when choosing and designing instruments. It also is important to distinguish between equilibrium costs and adjustment costs. Closing down old, polluting plants may be beneficial for the environment and, in many cases, for the economy overall. Closures still might cause considerable harm to employees and others in the form of short-run adaptation costs. The distribution of these damages between categories of people or between legal entities may vary just as much as the incidence of the environmental damages.

The cost distribution may have many dimensions—not only between owners of capital and laborers or between rich people and poor people, but also between generations, between the healthy and sick, between urban and rural, and among many other categories that may be difficult to enumerate or predict. Concern should be for the welfare of all citizens, and it is problematic that some groups are adversely affected because a particular characteristic is correlated with consumption or production patterns. People who live in a certain area, have a certain kind of family structure, job, culture, or habits, commute by a certain route or vehicle, or in some other way use certain goods may cause environmental damage; the policy instruments to correct their damage may end up imposing costs upon these same people. To a large extent, this effect is inevitable. Still, the extent of the costs borne by certain groups may sometimes cause problems.

Each case must be examined with an understanding that considerable welfare effects may be related to the exact distribution of costs and benefits and that these effects might even overshadow the aggregate effects. In some cases, the use of social welfare weights might be called for in the analysis, design, and evaluation of policy instruments. Particular care needs to be taken

BOX 12-2 Distributional Effects of Raising Carbon Taxes in Sweden

<i>Income group</i>	<i>Loss</i>	
	<i>SKr/cap/yr</i>	<i>% of total consumption</i>
Poorest 20%	888	1.24
Richest 20%	1,026	0.78
Urban areas	1,261	0.88
Rural north	1,392	1.16

Notes: Carbon taxes were increased from 0.37 to 0.74 Swedish krona [SKr] per kilogram (from about US\$0.04 to US\$0.07 per kilogram, where 10 SKr = US\$1). Loss is essentially measured as the compensation necessary to stay at the same utility level despite the tax. "Urban areas" refers to Stockholm and the two next largest towns.

Source: SOU 1997.

The results show distribution effects of increasing carbon taxes from 0.37 to 0.74 SKr/kilogram according to two criteria: income, and a combination of geography and rural/urban location. The richer consumers are more affected in absolute terms but less in relative terms. Judging this welfare effect is difficult because it depends on the curvature of the aggregate welfare function. The rural consumers of the cold and sparsely populated north who need energy for heat and mobility are, as expected, on average more affected than the inhabitants of the cities. However, it does not apply to all subcategories: Many rural farmers, pensioners, and others do not commute and use biofuels for heating. Thus, the issues are typically very complex. Total effects are quite small.

to avoid unnecessary effects related to unemployment and relocation. On the other hand, attempting to soften all the blows of economic restructuring can make policies very expensive. Good, detailed analyses of income distribution effects are rare. The Swedish Green Tax Commission is (at least partially) an exception (see Box 12-2).

One of the main purposes of the tax system is to alter (directly or indirectly) the distribution of goods and welfare. Public economics makes clear that taxing requires considerable information and runs the risk of creating incentive problems and control costs. Because of tax systems' complexity, adding new environmental goals exacerbates interaction effects and inevitably increases overall complexity. However, not only environmental tax instruments but also regulations have such interaction effects.

Poverty and Policy Instrument Selection

In developing countries, the poorest of the poor often suffer a disproportionately large burden of child mortality and disease. According to one study, the same people lack education as well as access to physical infrastructure such as water and sanitation: of all households, <10% of the third and fourth deciles,

30% of the second decile, and 70% of the poorest decile lack sanitary facilities (Bonilla-Chacin and Hammer 1999). Similar disparities apply to other indices, such as access to safe water (only 15% of the poorest decile). The difficulties are compounded because the poorest people often have low “social and human capital” resources. People with physical, mental, or social disadvantages (including India’s “untouchables” and refugees of natural disasters) are in this group. They may be unable to take advantage of programs intended to help them.

The political economy of instruments is complicated by the fact that lobby groups will use whatever argument they can—occasionally including untrue ones. Fuel taxes are of course opposed by both the oil industry and by professional users such as truck drivers. Lobbying campaigns often argue that a fuel tax would be regressive, hurting the poor more than the rich—a convenient argument whether or not it is true. Research shows that, especially in low income countries, such claims are far from the truth. Fuel taxes are actually quite strongly progressive in many low income countries (Sterner 2011). Nevertheless, they are not any easier to implement, because the governments of these countries are actually often more worried about opposition from urban elites than from the rural and poor populations.

One natural conclusion from this information is that poor populations should be better targeted, particularly in times of budget cuts and leaner government. However, it is far from easy to do because the “slightly less poor” have appreciable power to protect themselves from cuts in the benefits they receive, and it is possible (in some cases, even likely) that poor populations will be more than proportionately hurt by budget cuts (Ravallion 1999). A certain “leakage” to nontargeted (or less-targeted) groups may be part of the political economy cost of reforms; perfect targeting is simply not realistic considering the characteristics of these groups.³

At least some groups among poor segments of populations may be more than proportionately hurt by abatement costs and environmental programs; however, many poor people suffer severely from the consequences of natural resource degradation and certain types of pollution. This issue concerns not only the welfare but also the efficiency of poor households, inasmuch as their (potential) resources or abilities cannot develop given the seriously detrimental effects of a poor environment on their health and thus opportunity.

Supplemental Reading

Grafton and Devlin 1996
Grainger and Fullerton 2010
Fullerton and Heutel 2007

Kriström and Riera 1996
Parry et al. 2006
West and Williams 2004

Notes

- 1 In the context of the REPs, it would be hypothetically possible to emulate grandfathering by refunding the charges not to the current polluters but to polluters of some base year. However, it would be an odd policy to search out the largest polluters in some historic “base year” (even if they have closed down since then) and then pay what are essentially tax dividends from today’s polluters to them.
- 2 With taxes, Firm 1 will pay taxes $O'a_1XT$ and Firm 2 will pay $O'a_2WT$ (assuming O' is the origin from which emissions are measured). With subsidies per unit abated, Firm 1 will receive Oa_1XT , and Firm 2 will receive Oa_2WT (disregarding any output effects of the subsidies). With equal permit allocations of $O\hat{a}$, Firm 1 will sell $a_1XC\hat{a}$ permits to Firm 2, which is just sufficient for that firm (i.e., $a_2WC\hat{a}$).
- 3 This observation should not be taken as condoning complacency. In examples of seemingly successful reforms, general subsidies were cut and more targeted programs introduced; for example, in Tunisia, universal food subsidies were replaced with a food subsidy program that targeted poor individuals (Tuck and Lindert 1996). In Chile, strict marginal cost pricing was combined with means-tested subsidies for water pricing (see Chapter 23).

Politics and Enforcement of Policy Instruments

A common assumption is that policymakers design policies to maximize something like a social welfare function. Although this assumption is not very realistic, it is often made because it allows for the use of tractable mathematical models.¹ Even if policymakers did try to maximize such a function, and even if the welfare function were a simple utilitarian one, the design of policy might be complicated by market failures, information asymmetries, and other problems. If the social welfare function is more complex (e.g., including “equity” as an independent element), then the design of policy instruments must meet several goals. Also, complex individual utilities—for example, including altruism or relative (rather than absolute) consumption—have profound consequences for the design and implementation of policy instruments².

Some environmentally relevant goods may also be positional (e.g., car owners may get utility from having a better car than others), which will influence optimal taxation (e.g., Frank 1985). If there is habit formation, myopia, or even addiction, then the complexity of the utility function increases more (Rabin 1998; Becker and Murphy 1988), which may or may not have consequences for policy instrument design. If policymakers maximize a function completely different from welfare (e.g., chance of being reelected) or if they follow some other rationale for decisionmaking (e.g., psychological factors, ideology, or norms), then it must be incorporated into the analysis, as is traditionally done in the public choice literature—particularly if the purpose of the analysis is to be positive rather than normative. In this area, economists tend to recommend “optimal” policies (mainly based on efficiency criteria) and thus slip into a rather normative role. Equally as important as designing policies should be analyzing and understanding why certain policies are selected or designed differently under different circumstances. Psychological, cultural, and political factors may dominate over economic ones.

Political Economy of Policy Instrument Selection

To understand the politics behind the selection and design of policy instruments, one must realize that governance is a complex phenomenon. It is an area in which more collaboration is needed among economists, political scientists, psychologists, and other social scientists. Some political scientists categorize policy instruments as normative, suasive, or informative—or “carrots,” “sticks,” or “sermons,” according to Bemelmans-Videc, Rist, and Vedung (1998), who state that in some situations, the government must use “sticks” or “carrots,” but the tendency to use the more paternalistic informative instruments is increasing (see Chapter 4).

Political scientists have different views concerning the degree to which central government truly governs in the everyday sense of the word. Some contend that central government is being reduced to a position in which it coordinates and bargains with other centers of power, ranging from local government to supranational and intergovernmental organizations on the one hand, to networks of other actors in the private and public sectors on the other (Rhodes 1996). The opposite view is that the central governments maintain crucial control over the important decisions (Pierre and Peters 2000). Several theoretical models have been developed to analyze the political interaction of governments with various groups of interest in the setting and choice of regulatory instruments. One such conceptual construct is the median-voter model. In this setting, social choices are made by voters (or their elected representatives) and the final social outcome reflects the median of the most preferred outcomes of the individuals in the society. In the original version, this was basically a model of aggregation of demands for environmental regulation. Since it did not take account of the public sector supply, it was taken to imply that the final outcome is the most preferred by the median voter. Although subsequent research has shown that this result still holds in a setting where two political parties compete for voters (Downs 1957), the conditions under which the median-voter holds seem very unlikely. In addition, although the model provides some normative predictions concerning the need for environmental regulation, the explanatory power of the model becomes very limited when it comes to harder questions related to the stringency of the policies and the choice of policy instruments (Oates and Portney 2003).

The positive economic theory of regulation, initiated by Stigler (1971) and further developed by Peltzman (1976), sees regulation as a form of wealth transfer; regulation is not imposed on firms but demanded by them as a means to capture the regulator to favor regulations that restrict entry, support prices, or provide subsidies. In Stigler’s and Peltzman’s analyses interest groups (e.g., consumers and regulated firms) compete against each other in the political arena to achieve their objectives (such as environmental cleanliness and increased profits). That is, interests groups vie to shape regulatory initiatives in a way that will serve their own objectives. The state enacts the program of

whichever group offers the most resources to the government; regulation goes to the highest bidder, with the most powerful, well-established, and concentrated groups having an advantage over other groups. In the environmental area, a few industrial polluters often have more opportunity to band together as lobbyists than the much more numerous, dispersed, and unorganized victims of industrial pollution. In contrast, many polluters are unorganized and relatively powerless, and the capacity for nongovernmental organizations to capture and represent the interests of victims should not be overestimated.

Established industries have many strings to pull, not only through the traditional lobbying mechanisms but also through their relationships with employees, local communities, customers, and suppliers. Subsidies tend to be self-perpetuating in that their existence will be strongly defended by their beneficiaries, who may be prepared to spend a large share of the subsidy in lobbying for their continuation. On the other hand, individual licensing of plants is a process that can easily degenerate into a mechanism for limiting competition. For example, because technology improves constantly and because it is easier to incorporate new technology into new plants than to retrofit old ones, setting higher emissions standards for new plants might be considered a reasonable approach to regulation. When the old plants have lobbying power and succeed in lobbying for such a system, they effectively create a barrier to entry.

The choice of policy instruments is also affected by lobbying. A tax has the decided disadvantage of uniting polluters, who all lose money from a tax. Other market-based instruments, such as tradable permits and refunded emissions permits, may possibly diffuse some of the potential power of such lobbies because some segments of the polluting industry may gain rather than lose from the policies.

Conversely, firms that have political power typically have preferences about not only the target level of abatement but also the choice of instrument. Different instruments can have dramatically different consequences for the distribution of the total cost burden. The established firms have more influence and can shape technical regulations so that they have a heavy new source bias, which helps explain why physical regulations and licensing are so frequently used. Similarly, grandfathered permits are popular with established firms, whereas output-based allocation of permits and refunded emissions payments are less popular. Least popular, presumably, are taxes and auctioned permits. Striking a balance between what is perceived as fair while being realistic with respect to those who have power can be difficult. Usually, the success of a policy depends not only on a fair distribution of costs but also on respecting due process, which means following the traditional procedures for gathering information, engaging in debate, ensuring representation, and participating in decisionmaking. For example, Aidt (1998) finds that if the government seeks to maximize a mixture of political contributions and social welfare, competition among interest groups leads to an efficient internalization of detrimental externalities. If the interest groups represent the interests

of their constituencies faithfully, their contributions induce public decision-makers both to select efficient levels of externality-generating activities and to employ efficient regulatory instruments.

The political regime also affects the selection of policy instruments (Linder 1988; Linder and Peters 1990, 1991 and Lundqvist 2000). Democratic regimes tend to apply stronger policy instruments than authoritarian regimes as strategies for environmental abatement. Some countries have rules that restrict or prohibit lobbying, whereas others encourage or tolerate it; the prevailing approach affects the balance between lobbyists and technocrat experts arguing for efficiency, for example. The scope for local or regional instruments such as taxes or fees depends on their legality. Because many pollutants are either local or regional, policymaking ideally would take place at levels other than national. In many countries, local communities' rights of taxation are narrowly regulated, making it impossible to introduce environmentally differentiated waste charges, for instance. The state may be keen on keeping its monopoly on taxation, but more fiscal federalism and revenue sharing between levels of government often would be beneficial.

Similarly, national sovereignty sets limits to taxation and other instruments at the supranational or regional level. Another factor that influences instrument choice is the characteristics of the decisionmakers themselves. Rationality and time for decisionmaking are limited; decisionmakers cannot be expected to know everything about all relevant topics, and the time they are given to amass all relevant information on any given topic is usually short. As a result, they commonly rely on various methods to help them make their decisions: consulting experts, applying rules of thumb, learning by doing, and deferring to ideology. Ideologies are not trademarks invented to win elections; to a reasonable extent, they are belief systems that cover not only the goals of regulation but also the means, which is why some politicians prefer legal restrictions while others prefer financial incentives or the persuasive power of good examples.

In Part IV, we discuss how the political economy of lobbying can lead economies into distinctly different equilibria: some countries eschew fuel taxes almost completely and opt for physical regulation (e.g., the United States), whereas other countries, where the fuel taxes are much higher, only mildly oppose them and even tolerate further increases. One might expect that the political pressure to lower gas taxes would be high in countries with high fuel taxes, such as Sweden or Italy (see Chapter 17), and that it would be fairly easy to increase the modest gas tax in the United States. However, it is not: the parties who stand to lose are visible, organized, and vociferous, whereas those who potentially might gain are not organized and, in some cases, not yet in existence (e.g., people who would be employed if there were more public transportation options). Whereas the political resistance to a tax might be expected to increase with its magnitude, when it does not, it is indicative of a form of multiple equilibria or states that depend on the interaction between politics and economics.

Enforcement, Monitoring, and the Psychology of Instrument Choice

Traditional economic models analyzing compliance with environmental regulations assume that fines for law-breaking are just another cost of doing business. Polluters are assumed to act in order to minimize the sum of expected compliance costs plus expected penalties. If the compliance decision is binary, as in the case of a regulation requiring firms to install an indivisible item of abatement equipment, firms will choose to comply if and only if the cost of compliance is no greater than the expected penalty from noncompliance. However, if polluters can decide about the extent of the violation, and if the compliance cost increases with the level of compliance, the extent of the violation depends only upon the marginal properties of the penalty function. This principle, known as marginal deterrence, points out the importance of using a scheme of sanctions increasing with the extent of the violation. Failure to do so can explain apparent paradoxes of observed behavior—for example, situations where an increase in the level of penalties worsens compliance.

Environmental authorities can affect the structure of expected penalties by manipulating either inspection rates or nominal penalties or both. Since increasing the probability of detection requires some expenditure on monitoring, full compliance across the population might be achieved more easily by setting the penalty at some arbitrary high level. However, in many situations there is an upper bound on the penalty that can be levied. For instance, there could be legal limits to the size of penalties or political limits that try to prevent the bankruptcy of the violator. In other cases, the behavior of polluters creates a trade-off between the size of the penalty and the rate of compliance. Uncertain liability constitutes an example. Most analyses assume that if the regulator monitors and detects a violation, it can always impose a sanction without being challenged by the firm. In other words, once “detected” there is no uncertainty about liability. However, in reality a firm that violates the law might expend resources attempting to evade liability. Firms that are accused of violation might try to evade punishment by challenging the enforcement authority in court or bribing officials not to bring charges. These evasive activities complicate the basic optimal penalty result and the substitutability between the probability of detection and the size of the penalty. Instead, to increase compliance, the policy prescription is to lower the penalty and increase the government monitoring efforts.

Self-reporting is another example of a situation where there is an upper bound on the penalty that can be levied. If firms are compelled to self-report their emissions, the enforcement strategy must address two types of violations: emissions violations, where firms fail to fulfill the law, and reporting violations, where firms do not give an accurate report of their emissions. Setting a very high penalty for noncompliance may deter violations, but it also provides an incentive for firms to evade detection and to reduce the accuracy of self-reported emissions. This indicates an essential trade-off between encouraging firms to reduce pollution and encouraging them to report emissions honestly. By lowering the

finest for noncompliance, the regulator buys information on the true compliance status of the firm (Cohen 1999; Sandmo 2002).

Although the welfare gains from self-reporting are ambiguous (Malik 1993), environmental agencies heavily rely on self-reporting by sources³. Kaplow and Shavell (1994) argue that self-reporting schemes can improve upon random inspection schemes because they lower the costs of monitoring and auditing firms. If self-reporting speeds up the process by which victims are notified, it might also help to reduce the damages caused by pollution since mitigating actions can be taken sooner to avoid further harm. Burlando and Motta (2008) highlight the role self-reporting plays in reducing corruption. Without self-reporting, the environmental authority can induce officers to refuse bribes that are offered to them only by providing bonuses to those officers who indeed report violations. In order for the bonuses to be effective, they must be of a higher value than whatever bribe a firm in violation is willing to offer; a bidding war ensuring high payments to the officers ensues. Self-reporting provides an alternative strategy to the firm in violation: it can now simply avoid paying the bribe by paying directly to the government. When the firms in violation choose this alternative, they avoid the confrontation with the officer, who cannot claim either bribe or bonus (or at least, it sets a cap on bribes).

The use of differential monitoring rates and differential penalties—based on prior compliance history or other exogenous information—can help to reduce government enforcement expenses. Harrington (1988) models the regulatory attempts to minimize the resources devoted to monitoring and enforcement, consistent with ensuring a minimum rate of firms in noncompliance and with fines not greater than a given maximum. He uses a repeated enforcement/compliance game, where the regulator groups firms according to recent inspection history: Group 1 consisted of individuals found in compliance at their last audit, and group 2 consisted of individuals found in violation when last audited. The regulator can modify several variables: the two monitoring probabilities, the two penalties, and the probability of allowing firms in group 2 back into group 1 after being found in compliance. In the simplest case, the regulator levies no penalty upon a group 1 firm caught violating but a maximal penalty upon a group 2 firm caught in violation. He finds that social cost is minimized when the monitoring probability for group 1 is lower than that for group 2 and that in equilibrium a representative firm can be induced to comply a significant fraction of the times despite penalties never actually being levied. This is the so-called Harrington paradox, namely that firms comply most of the time despite the fact that when the regulator observes violations it often chooses not to pursue the violator and that the expected penalty faced by a violator is small compared with the cost of compliance.

Harrington's results suggest that a regulator who agrees to certain forms of flexibility with cooperative firms acquires a great deal of power or leverage vis-à-vis the uncooperative firms (however, this mechanism is almost the same as the one corrupt officials use to collect bribes). In this kind of model,

overcompliance may well be the optimal response. Nevertheless, overcompliance has several other plausible explanations. For example, it could stem from the (altruistic) preferences of managers as individuals or from a misjudgment of the risk for (or consequences of) being monitored and prosecuted. These risks are small, but managers may be averse to these kinds of (partly private) risks. Detection and prosecution for environmental noncompliance may carry more costs than the official fines or other penalties; the company's reputation and trademark could be tarnished, employee motivation could suffer, and shareholder interest could decrease in response to negative publicity.

The random nature of the pollution's discharges constitutes another alternative explanation for overcompliance. Bandyopadhyay and Horowitz (2006) show that plants with greater discharge volatility had lower average discharges, which suggests that firms may hedge to provide a margin of safety against violations due to stochastic discharges. When stochastic shocks are particularly large, firms may reduce their average discharges in an effort to stay compliant.

Theoretical models by Arora and Gangopadhyay (1995), Kirchhoff (2000), and Cavaliere (2000) show that consumer preferences for environmental quality can also generate overcompliance as an outcome of noncompetitive markets. Publicly available information on environmental performance of firms enables consumers to identify clean firms and leads some firms to overcomply in order to obtain market differentiation. These models also help explain why environmental abuse by firms in developed countries can be less than that of their counterparts in less developed countries. Higher income levels in developed countries have increased demands for environmental quality, forcing companies to address environmental concerns more seriously.

When formal regulation is weak or absent, communities often use other channels to induce pollution abatement by local firms in a process of informal regulation. Hettige et al. (1996) explore the role of community pressure in the degree of compliance with environmental regulations. They find that the ability of communities to play this role appears to be an increasing function of their income and education level.

Usually, economic incentives are assumed to have an effect on actions (strong or weak, but definitely positive). However, economic incentives can "crowd out" moral motives (Frey 1997). In one experiment, two groups of subjects played managers who were instructed to allocate part of their budget to scrubbers that would reduce emissions (Tenbrunsel and Messick 1999). One of the groups was required to make decisions subject to a weak external sanction: noncompliance led to a small risk that the group would be fined. Compliance was about 75% in the reference group but 50% in the "sanctioned" group. Without the sanction, the decision was seen as an ethical one; with the condition, it became a business decision, and because the expected costs of cheating were low, abatement was simply not profitable⁴. In a similar study that focused on common property resource management, people who used the same finite water resource were offered the opportunity to voluntarily "buy out" other users to save the

resource—i.e., a smaller group would more easily think of the common good (White et al. 1995). However, instead of the intended result, the policy hastened depletion. In debriefing, participants indicated that the payment structure made them think in terms of maximizing their own outcome rather than solving the collective dilemma. Thus, economic incentives had an effect that was perverse or opposite to that intended.

There is no neat way of summarizing the above factors, but their influence on instrument choice or design can be profound. In a culture where polluters dutifully follow regulations despite insignificant penalties, inefficient monitoring, and lax enforcement, simple policies may be sufficient. In other cultures, such a policy would be overly naïve. Considering the social value of compliance, policymakers should consider the long-run effect of new regulations (and policy instruments) on public attitude. Unreasonable legislation, for instance, may weaken a law-abiding culture in general.

Enforcement and the Choice of Policy Instruments

Analyzing how the different regulatory instruments work under imperfect compliance is important in order to determine which policies to choose. To perform such a ranking, several aspects need to be considered: (1) the type of pollution; (2) the set of policy instruments available; (3) the behavior of polluters, and (4) the motivation of the enforcement agency.

With regards to enforceability, several characteristics of emissions influence the choice of policy:

- some emissions are observable to the auditor⁵ (point sources discharges) while others are unobservable (nonpoint sources);
- some emissions come from the intentional decision to violate the law while others are stochastic by nature (in such a way that it is not easy to know whether the existence of pollution is due to willful violation of the law, some negligent behavior, or a random act of nature); and finally,
- some environmental problems come from a few big agents while in other cases the emissions come from many small agents.

It seems intuitive that pollution that is very difficult to monitor or that involves many small agents is easier to control indirectly, via taxes on the input or on the output of polluters. Direct controls are easier to implement in those cases where pollution is observable and comes from some big and few agents that can install abatement technology to reduce emissions. For other cases, the regulator may try to build a culture of compliance via, e.g., green labeling, to affect consumer behavior or investor decisions. In all these cases monitoring is still a concern. Studying imperfect compliance allows a better design of environmental policies.

Although enforcement issues seldom drive the choice of regulatory instruments⁶ theoretical findings have shown that there is a fundamental asymmetry

in the nature of firms' responses to imperfectly enforceable versions of standards and taxes. If the only objective of the environmental authority is to minimize total emissions at the minimum cost, environmental taxes are more effective than standards. The key of this result is the recognition that a pollution tax is essentially a fine for violating a standard that allows zero pollution (Box 13-1).

BOX 13-1 Deterrence Under Emission Standards and Emission Taxes

Let us consider the case of a non random pollutant where monitoring is costly but allows the environmental authority to determine the firms' emissions. The firm's benefits from emission e are represented by the function $g(e)$ that is increasing and concave $g'(e) > 0$ and $g''(e) < 0$ for all $e > 0$. Firms have the obligation to self-report their emissions. If the level of emissions is not perfectly and costlessly monitorable, the monitoring and enforcement strategy and the reporting strategy of the firms are strategic decisions. Thus, firms may choose to report a level of emissions z that does not coincide with the true emission level e . When the firm reports a level of emissions higher than the standard, it has to pay a fine τ per unit of excess emissions. If the firm underreports and if it is audited and its true emission level is identified, it has to pay a penalty that depends on the extent of the under-reporting. Let α denote the probability that the environmental authority audits the firms. The difference between reported and true emissions is known as "violation." The penalty takes the form of a function $\theta(e - z)$ increasing and convex in the level of violation.

A firm chooses the optimal levels of emissions in order to maximize the expected profits net of compliance and policy related costs. If the regulator sets a standard, expected profits are given by:

$$E\pi = g(e) - \tau(z - \bar{e}) - \alpha\theta(e - z) \quad (13.1)$$

where $g(e)$ is the benefit from emissions, τ is the fine, z is the level of emissions reported, $\bar{e}$ is the emission standard, α is the probability of being audited and $\theta(e - z)$ is the penalty.

The following conditions hold:

$$g'(e) = \alpha\theta'(e - z) \quad (13.2)$$

$$\tau = \alpha\theta'(e - z) \quad (13.3)$$

Condition (13.3) says that the firm chooses its level of reported emissions such that the fine equals the marginal expected penalty. If the firm is not subject to any audit ($\alpha = 0$), it pollutes freely until the marginal benefit of emissions is equal to zero. As the audit pressure on the firm increases, the firm decreases its level of emissions until the marginal benefit of emissions equals the marginal penalty.

If the regulator uses taxes instead, the expected profits of the firm are given by:

$$E\pi = g(e) - Tz - \alpha\theta(e - z) \quad (13.4)$$

where T is the tax per unit of emissions. The following conditions hold:

$$g'(e) = \alpha\theta'(e - z) \quad (13.5)$$

$$T = \alpha\theta'(e - z) \quad (13.6)$$

Condition (13.6) states that firms choose to report a level of emissions such that the marginal expected penalty equals the tax.

Comparing (13.1) and (13.4) it can be seen that they coincide with each other when $\tau = T$ and $\hat{e} = 0$. If $\hat{e} > 0$ and for the same probability of being audited α , the environmental authority is able to induce a lower level of pollution with taxes than with standards. Intuitively, standards reduce the size of the penalty because they increase the minimum report firms may do without cost. Only when the penalty function is linear do standards and taxes lead to the same results.

Thereby, the environmental authority is able to induce a lower level of pollution with taxes than with standards since the relative size of the economic penalty and the marginal deterrence are larger.

As mentioned above, the use of differential monitoring can help to reduce government enforcement expenses. Macho-Stadler and Pérez-Castrillo (2006) study the optimal audit policy for an enforcement agency whose objective is to minimize the level of emissions with emission taxes. They show that it is optimal to devote the resources primarily to the easiest-to-monitor firms and to those firms that value pollution less. However, unless the budget for monitoring is very large, there are always firms that do not comply with the environmental objective.

Enforcement in an emission trading system confronts regulators with a more complex enforcement problem than with taxes and performance standards since they must focus not just on emissions but also on the behavior of firms in emission permit markets (Stranlund and Chavez 2000). To detect noncompliance, the regulator must keep track of the number of permits firms hold and ensure that firm emissions do not exceed that number of permits. Finally, the authority must be able to enforce compliance by penalizing firms misreporting or emitting in excess of the permits they hold. Effective enforcement of emission trading systems with self-reporting requires penalties to be tied to the permit equilibrium price (Box 13-2). To understand this relationship, consider the case of a firm in an emission trading program. The firm will choose how much pollution to emit and report and how many permits to hold in order to minimize the expected costs, which consist of emission control costs, receipts or expenditures from permit market transactions, and expected penalties from reporting and emission violation. Assume that the permit price in a competitive market is given by p , the probability that the environmental authority audits the firm is α , the per unit fine levied for emission violation is τ , and the penalty levied for reporting violations is given by the function $\theta(e - z)$, where e and z denote actual and reported emissions, respectively. Two

BOX 13-2 Deterrence Under Tradable Permits

If the regulation is based on tradable permits, L permits are allocated among the firms. Then, the expected profits of the firm are given by:

$$E\pi = g(e) - p(l - l_0) - \tau(z - l) - \alpha[\theta(e - z)] \quad (13.7)$$

where in this context, l denotes the number of permits that the firm holds in equilibrium and l_0 denotes the number of emissions permits initially allocated to a firm. The following conditions hold:

$$g'(e) = \alpha\theta'(e - z) \quad (13.8)$$

$$\tau = \theta'(e - z) \quad (13.9)$$

$$p = \tau \quad (13.10)$$

If the regulator wants to have complete compliance, that is, $e = l = z$, there are two conditions that must be satisfied: $p < \alpha\theta'(0)$ and $p < \tau$. The first condition ensures that firms will provide truthful reports on their emissions. The second condition ensures that the penalty for emissions violations is higher than the permit price—that is, it is higher than the cost of being compliant.

conditions must be satisfied in order to induce complete compliance: (1) $p < \alpha\theta'(0)$ and (2) $p < \tau$.

The first condition ensures that firms will provide truthful reports on their emissions. To understand this, notice that when a firm decides to misreport a violation, it gains the foregone cost of being in compliance, which is the permit price. The expected marginal cost of misreporting emissions is given by the marginal penalty $\alpha\theta'(e - z)$. Hence, a firm will provide a truthful report of its emissions ($e = z$) when the marginal benefit of underreporting its emissions p , is not greater than the expected marginal cost of doing so $\alpha\theta'(0)$. On the other hand, the permit price also represents the marginal benefit of noncompliance. Given the proper incentive to provide truthful emissions report, a firm that is in violation will choose complete emission compliance if the price of being compliant p is less than the certain marginal penalty for emissions violation τ .

Thereby, the enforcement strategy required to maintain complete compliance strongly depends on the prevailing market price and not specific features of the firm. However, in imperfectly competitive markets, firms can exercise market power, and their compliance incentives will also depend on the extent to which the firms can manipulate permit prices (Chavez and Stranlund 2003). On the other hand, risk aversion might reduce the extent of violations; noncompliant firms with risk-averse managers will have lower violations than otherwise identical firms with risk-neutral managers. In equilibrium, market

risk aversion will be associated with higher permit prices and better environmental quality (Stranlund 2008).

In practice, some fraction of firms underreports emissions and does not comply. We can think of at least three reasons for this. First, it may be that the probability of getting caught in violation is heterogeneous, varying from firm to firm. When this is so, only firms with a high probability of being audited choose to fully report their emissions. Second, a firm may have more to hide than just the emissions violation. Self-reporting emissions violations may lead government to audit other aspects of the firm more thoroughly. Finally, firms face uncertainty due to the complexity of the laws. They cannot be sure whether self-reporting a violation will cause them to be held accountable for other acts. Whenever these circumstances arise in a way that cannot be priced in the penalty, some firms will underreport, preventing the environmental authority from inducing perfect compliance.

Under imperfect compliance, the expected costs of environmental policies include not only the firms' aggregate abatement costs and the government's monitoring costs, but also the expected costs of sanctioning noncompliant firms. Stranlund (2007) analyzes the choice of penalty structure in emissions trading schemes. In his setting, a positive amount of noncompliance is cost-effective if violations are punished with an increasing marginal penalty. The optimal level of noncompliance depends on penalty collection costs, monitoring costs, and the parameters of the penalty function. If the marginal monitoring costs are relatively high, then the solution calls for a certain amount of noncompliance to conserve monitoring costs. Instead, if marginal sanctioning costs are high, then the solution calls for more intense monitoring to induce perfect compliance to eliminate the expected costs of collecting penalties.

In a recent paper, Villegas-Palacio and Coria (2010) analyze the effects of the interaction between technology adoption and incomplete enforcement on the extent of violations and the rate of abatement technology adoption. This interaction can be thought of in two ways: (1) incomplete enforcement, and therefore the possibility that firms do not comply with a regulation, may influence the profits of firms' technology adoption choices (see Chapter 9), and (2) the existence of a new technology that reduces the abatement costs may influence a firm's compliance decisions, since the marginal benefit of violations is reduced. They show that when taxes are used, the rate of technology adoption is not affected by enforcement strategy. In contrast, when tradable permits are used, the rate of adoption is an increasing function of the monitoring probability. Additionally, under tradable permits, for a given monitoring probability the fall in permit price produced by technology adoption reduces the benefits of violating the environmental regulation at the margin, ultimately leading both adopters and nonadopters to modify their compliance behavior. This is not the case under emissions taxes, where the tax rate remains unchanged after technology adoption. Thus, in contrast to taxes, the extent of violations under tradable permits decreases with the rate of adoption. Moreover, when tradable

permits are used, the deterrent effect of the monitoring effort on violations is reinforced by the effect that technology adoption has on the equilibrium permit price and consequently on the extent of violations.

Lately, interest has been growing in the use of voluntary agreements between regulators and polluting firms in place of standard regulatory approaches (see Chapter 8). Although voluntary agreements are thought to have a number of advantages over traditional instruments, they also might be subjected to firms' manipulation. For example, a firm may be motivated to join an agreement to help prevent the imposition of some conventional regulation but then decide not to reduce its emissions to the extent required under the agreement. McEvoy and Stranlund (2010) examine the consequences of costly enforcement on the ability of voluntary agreements to meet regulatory objectives, the levels of industry participation with these agreements, and the relative efficiency of voluntary approaches. Their results suggest that, under a limited set of circumstances, voluntary agreements can be more efficient than emission taxes since the incentives to free ride on the agreement are reduced. For instance, when the members of the agreement bear the enforcement costs, when a third party enforces it, or when the enforcer of the agreement has a significant advantage over the government in terms of its ability to monitor firms' compliance or in terms of the sanction, it can apply in cases of noncompliance.

Some studies use laboratory experiments to examine compliance behavior under uniform taxes or tradable emissions permits. For example, Cason and Gangadharan (2006) use laboratory experiments to identify interactions among emission shocks, banking, and compliance and enforcement in an emissions trading market in the presence of emissions uncertainty. They find that banking helps smooth out the price variability from the imperfect control of emissions; however, noncompliance and emissions are significantly greater when banking is allowed. Murphy and Stranlund (2006) study the direct effects of enforcement on compliance and emissions decisions as well as the indirect effects that occur due to changes in permit prices. They find experimental results consistent with theoretical predictions about a negative direct effect of enforcement on individual violations and a counteracting effect through the permit price. In a related paper, Murphy and Stranlund (2007) test the hypotheses that both the violations of risk-neutral firms and the marginal effectiveness of increased enforcement across firms are independent of differences in their abatement costs and their initial permit allocations. Finally, Stranlund et al. (2008) investigate whether gross profit, aggregate expected penalties, and aggregate expected net profits deviate from theoretical predictions based on risk-neutral profit-maximizing firms with imperfect enforcement. Their findings suggest that observed violations in the experimental setting are lower than the predicted levels when subjects have strong incentives to violate their emission permits, and that individual emissions control responsibilities are distributed among firms as predicted.

Policymaking in Severely Resource-Constrained Economies

Examples of the Harrington paradox are also found in developing countries. Still, in some contexts, compliance appears to be lower, but whether it depends on the weakness of laws, public agencies, or other factors is unclear. Because monitoring institutions are weak overall, it is impossible to know for sure whether compliance is common (see for example, Box 13-3).

In poor countries, many social, political, and other factors distinctly different from those in developed countries must be taken into account in designing policy instruments. Poverty is often related to several factors that make policy design and implementation difficult: low education, lack of finance, lack of administrative tradition, and weak institutions. With the tensions present in some of the multiethnic nations carved out by the colonial powers, these factors sometimes contribute to higher-than-usual incidence of corruption and nepotism. Corruption has been identified as a major threat to development in general (Keefer and Knack 1997) and makes the design of policy instruments difficult in particular. Judicial corruption and favoritism can lead to distortions in agents' incentives to invest in assets such as trust and reputation and thus reduces the efficiency of policy instruments (Mui 1999).

Other special circumstances typically apply in developing countries: the informal sector (e.g., small or home-based businesses) is large, making monitoring difficult; property rights may be poorly delineated (often as a result of conflicts between many layers of historical and cultural traditions); agriculture, fishing, forestry, and other industries based on often-sensitive ecosystems tend to be the mainstay of the economy (Platteau 1999); and capital and credit markets are marked by risk, uncertainty, and restrictions, making it difficult to borrow money to finance "nonproductive" (e.g., abatement) investments.

BOX 13-3 Trading Programs and Compliance

Permit trading has become a popular policy instrument to achieve emissions reductions during the last decades. A number of programs at the firm level as well as national and international levels trade permits for PM, SO₂, NO_x and CO₂. Theory suggests that to induce perfect compliance, penalties for violation must exceed the permit price. This conceptual explanation of emission trading programs incentives explains the perfect compliance record in the U.S. Acid Rain trading program. From 1996 to 1999, allowance prices in this program ranged from US\$ 68 to over US \$200 per ton while penalties were US\$ 2,581 per ton of excess emissions. Noncompliant utilities also had to offset excess SO₂ emissions from the next year's allowances allocation (Stranlund et al, 2002).

Now look at the large rate of noncompliance in the PM trading program in Santiago, Chile, which ranged from 22% to 46%. In 2005 the PM price was US\$ 5,230 per kg while actual sanctions were decided on a case-by-case basis and ranged from US \$4.50 to US \$90,000 (Coria and Sterner 2010).

Frequently, abatement costs are not annual variable costs but investment costs. Liquidity problems result because abatement equipment incurs high costs at the same time that the polluting firm is expected to pay its unusually high taxes⁷. This situation can be alleviated by using tax or charge proceeds to create environmental funds for financing loans to polluting companies for abatement-related outlays; such funds also can finance local environmental protection agencies (Panayotou 1995, 1998). Liquidity problems can also be lessened by refunding emissions payments or by using a two-tiered tax (see Chapters 6 and 7) such as the tax–subsidy scheme, whereby polluters pay taxes only on “excess pollution.”

The technology and ecology of an issue may be so complex that individual licensing must be part of the policy package chosen. Unfortunately, licensing creates risks of “regulatory capture”: polluters dominate (and maybe corrupt) the regulators. Polluting firms typically have more information and greater resources at their disposal than regulators, who face the dilemma of not only designing policy instruments but also—first and foremost—constructing their own agencies. Together with state and local inspectorates, U.S. EPA employs on the order of a hundred thousand individuals; most agencies in developing countries have only a few employees, who sometimes have little technical training and equipment as well as limited legal authority. Managing the environment and natural resources is no trivial task for a handful of civil servants. However, training, dedication, and other factors are probably more important than sheer numbers.

Under these less-than-ideal circumstances, informational policy instruments may be a good first step in regulation. By collecting and disseminating information, an environmental protection agency can achieve several important goals:

- creating a baseline for future action;
- encouraging transparency so that individual inspectors can at least not secretly agree to unreasonable emissions;
- building relationships with the polluting industries that are not purely adversarial; and
- opening the way for pressure from customers, workers, investors, neighbors, and other involved parties.

Many (but far from all) problems can be greatly diminished by simple means, and positive partnerships must be built in which the environmental protection agency may assume the roles of not only “police” but also “facilitator” and “teacher.” Local industries must acquire a reasonable perspective on the range of abatement technologies as well as the requirements posed by customers in national and international markets. Local agencies cannot perform this task or any other (e.g., facilitating contact with credit institutions) well if they are understaffed and underfunded. Small fees on appropriately selected pollutants may help provide required financing.

A certain degree of pragmatism is important here. Finding the exact Pigovian level of a certain tax may not be important in an economy in which the level is likely to change by several orders of magnitude as a result of rapid industrialization, migration, technical progress in abatement, and other factors related to resource-based or informal-sector activities and poorly developed markets and institutions. However, information policies may be treated as a first step toward fees and other instruments, such as labeling (e.g., see Chapter 21 on PROPER). Recording information early in the process is important as a baseline against which to evaluate the success of policy measures undertaken. Without such evaluations, the local environmental protection agency will not be able to learn from its successes and failures.

The social and political (as distinct from economic) costs of bankruptcy can be high, which tends to indicate that high taxes are hard to collect. Establishing the principle that pollution is a cost to be internalized is crucial, and a low fee that is implemented and leads to payments (and some incentive effects for abatement) is better than a fee that has no chance of being implemented. If carefully designed, a charge that is partly refunded to the industries to finance abatement and partly used to finance the environmental protection agency may bring all parties into a productive collaboration. (Although risks are involved in this arrangement, most developing countries that use environmental charges also put the proceeds into environmental funds—see Chapter 21). One of the advantages of such a fee is that it may be set considerably higher than a pure tax.

Supplemental Reading

Pollution Policy in Transitional/Developing Countries

Bluffstone and Larson 1997
 Carlsson and Lundström
 2000
 Coria and Sterner 2010
 Vincent and Farrow 1997

Psychological Decision Theory

Montgomery 1983

Public Choice

Arrow 1951
 Buchanan and Tullock 1962
 Tullock 1965, 1981

Political Economy of Environmental Taxes

Aidt 1998
 Boyer and Laffont 1996
 Dijkstra 1999
 Keohane, Revesz, and Stavins 1998
 Oates and Portney 2003
 Wallart 1999

Environmental Policy and Imperfect Compliance

Cason and Gangadharan 2006
 McEvoy and Stranlund 2010
 Macho-Stadler and Pérez Castrillo 2006
 Sandmo 2002
 Stranlund 2007
 Villegas-Palacio and Coria 2010

Notes

- 1 Actually, ecologists also assume that plants, animals, and even genes maximize something akin to utility: “fitness.” This practice does not necessarily reflect a belief in the existence of conscious purpose in nature but reflects the fact that if maximizers compete with nonmaximizers, then maximizers will tend to win and thus dominate in the long run through natural selection.
- 2 For an analysis of environmental taxation under different forms of altruism, see Johansson 1997b. The debate on whether utility is derived from mainly absolute or relative income has many antecedents in economic thinking. The mainstream emphasis is on absolute income, but some authors (e.g., Oswald 1998) conclude that relative income is dominant (for an overview, see Weiss and Fershtman 1998). Johansson-Stenman, Carlsson, and Daruvala (2002) carry out an empirical test and find both to be important.
- 3 Russell (1990) found that, on average, state environmental agencies in the U.S. required 28% of air pollution sources and 84% of water pollution sources to self-report.
- 4 The financial incentive was designed to be weak; a strong incentive might have reinforced cooperative decisionmaking. Some research finds that with higher financial stakes, behavior tends to be “expected” and “rational” (Smith and Walker 1993).
- 5 The categorization of some pollution sources as “point” sources and others as “nonpoint” sources reflects the capability of existing technology for emissions monitoring and other factors such as the cost of the technology. The classification of an individual source of pollution may change over time as monitoring technology advances and the costs of monitoring decline.
- 6 This might not be true in some developing countries, if the large informality of the economy makes the cost of monitoring too high to implement emission fees or direct controls.
- 7 The taxes are high mainly because the firm’s emissions are high. With high total emissions, the marginal damage is also high. The literature is somewhat ambiguous as to whether the appropriate Pigovian tax is the marginal damage in equilibrium or in the current situation (Baumol and Oates 1988, Chapter 6.7). With the latter interpretation, the tax rate would be much higher, compounding the liquidity problems for the firm.

International Aspects and Climate Change

Governments are bound by numerous international conventions concerning the interplay of the environment and natural resources with investments, political relations, and world trade. At least two distinct sets of factors affect domestic environmental policymaking imposed by international regulations. On the one hand are concerns over the effects of environmental policies on international competitiveness. On the other, international policymaking plays a necessary role both in providing public goods and dealing with transboundary or global pollution, such as the greenhouse gases contributing to climate change. In this chapter, we first explore the interaction of environmental regulations and international trade and then analyze transboundary pollution and climate change.

International Trade and the Environment

International trade has been central to economic thinking for well over two centuries. Beginning with the seminal work of Adam Smith and continuing with the work of David Ricardo, economists have considered international trade to be a source of many gains. For instance, the notion of comparative advantage has been used to demonstrate that two nations can trade to their mutual benefit even when one of these nations is more efficient than the other at producing everything. Second, trade allows nations to export goods whose production makes relatively heavy use of resources that are plentiful nationally while importing goods whose production makes relatively heavy use of resources that are scarce nationally. Finally, trade allows nations to specialize in producing a narrower range of goods, thereby permitting them to enjoy the greater efficiencies of large-scale production.

Despite the many potential gains from international trade, in recent times the desirability of free trade has been questioned due to the possible adverse impacts on the environment. Environmental groups fear that a more

integrated world economy will hasten the exhaustion of natural resources, increase air pollution around the world, and encourage polluting industries to move to countries with lenient environmental regulations. In this line of thinking, the *pollution haven hypothesis* states that given the pronounced differences in the level of stringency of environmental policies between developed and developing countries, the industries that are highly pollution intensive will migrate from the former to the latter. An interjurisdictional competition may lead to a *race to the bottom*: since all governments are interested in attracting mobile factors of production to their own nations and complying with stringent environmental regulations is costly, governments may deliberately weaken their environmental regulations to attract these mobile factors of production.

On the other hand—and less widely recognized—is the possibility that international trade could have a positive effect on environmental quality for a number of reasons. First, if trade raises income, it allows countries to attain more of what they want, which includes environmental goods as well as more conventional output. Second, trade can spur managerial and technological innovations and the spread of good practice, which can have a positive effect on both the economy and the environment. Indeed, multinational corporations tend to bring clean—state of the art—production techniques to host countries. Third, tougher environmental standards can spread through lobbying and heightened public awareness. Whereas some environmental gains may tend to occur with an increase in income, others may be more likely when associated with international trade and investment. Much of this was described by Porter (1991) and Porter and van der Linde (1995), who state that well-designed regulation can promote competitiveness. This idea derives from a more dynamic view of competitiveness, which gives a central role to technological change; regulation signals firms about likely resource inefficiencies and potential technological improvements, it increases corporate awareness, and it creates pressure that motivates innovation. The result is in many cases a process that not only pollutes less but also lowers costs or improves quality. Thus, firms introducing more environmentally friendly technology may enjoy a “first mover advantage” vis-à-vis laggards who continue to use traditional production methods (see Ambec and Lanoie 2008; Ambec et al. 2010).

Whether the race to the bottom effect dominates the gains from trade is an empirical question, and the available evidence is ambiguous. Although the theoretical justification for the pollution haven hypothesis is strong (see Pethig 1976; McGuire 1982; Krutilla 1991), most existing empirical studies have failed to find a robust effect of weaker or stricter environmental regulations on industrial location (Xing and Kolstad 2001; Eskeland and Harrison 2003). One stumbling block of this analysis is the difficulty in quantifying whether environmental legislation is lax or strict. Another is finding a counterfactual situation. Such trends presumably would take a long time

to materialize, and during that time, many other important factors at play are related to the laxity of legislation and the patterns of growth. Results are difficult to identify statistically.

In addition, the pollution haven hypothesis presumes that environmental cost differentials are a sufficient inducement for a firm to relocate its production process; however, many studies have shown that the share of pollution abatement in operating costs is relatively low, although the proportion of investment devoted to pollution control is usually much higher. The impact of investment in pollution control equipment on competitiveness depends on the ability to pass on those costs to consumers in the form of higher prices when polluters are compensated for the costs of environmental protection through subsidies, import restrictions, or tax concessions. This neutralizes the effects of regulations on competitiveness (see Eliste and Fredriksson 2002).

The debate on Porter provides an interesting perspective on scientific debate. The original articles by Porter were fairly general and multifaceted. One of his many statements was that environmental regulation may spur technical innovation, in turn partly (or even fully in some cases) offsetting the extra costs of regulation—an idea known as *the Porter hypothesis*. But if it is taken to mean that “environmental regulation increases productivity,” this caricature has failed to find theoretical or empirical support. Such an oversimplification has been criticized by economists who say that if the productivity opportunities were real, then they would be exploited irrespective of legislation. Xepapadeas and de Zeeuw (1999) use a dynamic model of the firm to show that a more stringent environmental policy reduces capital stock (downsizing), and reduces the average age of the machines (modernization). These effects result in an increase of average productivity and profits reducing the trade-off between environmental conditions and profits. Simpson and Bradford (1996) evaluate the argument that stricter regulation will enhance domestic competitiveness in a strategic trade model. They find that although this may be a theoretical possibility (especially if domestic firms can exercise market power, domestic and foreign products are differentiated, or innovations can be licensed), using environmental regulations as industrial policy is dubious as practical advice. As they point out, maybe the best argument in favor of the use of environmental policy—instead of other industrial policies to increase innovation expenditures—is that it would not encourage manipulation or entry: innovation subsidies might be sought by firms while stricter environmental regulations would not. However, freedom from manipulability seems a poor reason to advocate for more stringent environmental policies to advance industrial policy objectives.

Maybe it comes as no surprise that a country’s choice of environmental policy instrument might also affect the international competitiveness of its firms. Sartzetakis and Constantatos (1995) show that the total market share of firms regulated through a tradable emissions permits (TEP) system increases relative to one where firms operate under a command-and-control system

because a TEP system better allocates total abatement among firms. This result suggests that in the long run, free trade situations might result in similar regulatory regimes.

Environmental Regulations as Strategic Trade Instruments

While many groups in developed countries are concerned about industrial flight and the pollution haven hypothesis, some developing countries expressed concerns about a “new protectionism” that uses trade measures like green labeling and import bans to achieve environmental goals. At the heart of all these concerns is the fact that environmental regulations may be used as strategic trade instruments, especially in the context of imperfectly competitive international trade. The term “strategic environmental policy” refers to environmental policy oriented toward promoting exports rather than abating pollutions.

Conrad (1993), Barret (1994), and Kennedy (1994) were the first to discover that in the presence of imperfect competition (Cournot), emission taxes can be used to indirectly subsidize exports by underinternalizing environmental damages. They consider environmental policy in a conventional strategic trade framework in which a polluting industry in each of two countries competes to export output to consumers in a third country. The outcome of these models has become known as the “eco-dumping” effect: National regulators have an incentive to set taxes below the Pigovian level in noncooperative trade contexts. Several authors have extended these results by including imperfect information, pre-investment in cost reduction, R&D, or choice of location (see Requate 2005b). To analyze the structure of the unilateral second-best optimal tax, let us introduce a simple model. Assume two countries, one domestic (D) and one foreign (F), that due to a *free trading agreement* are not allowed to impose tariffs to subsidize their firms directly. The world inverse demand function and the domestic demand function are given by $P_w(Q_w)$ and $P_D(Q_D)$ where Q_w is world total output and Y_D is the domestic level of consumption. There are n_D domestic and n_F foreign firms. Within the domestic country, all firms have identical technology. $C^D(q_D, e_D)$ represents the cost function of a typical domestic firm. Domestic damage is $D^D(E^D)$, where $E^D = n_D e_D + s_F^D n_F e_F$ is pollution in the domestic country caused by emissions from both domestic and foreign firms. Foreign emissions are multiplied by the transfer coefficient s_F^D which indicates how much foreign emissions are deposited in the domestic country. If we assume firms compete à la Cournot in an international market that may also consist of third-country markets, social welfare in the domestic country is given by:

$$W = \int_0^{Y_D} P_D(Y_D) dY - n_D C^D(q_D, e_D) + P_w(Q_w)(n_D q_D - Y_D) - D^D(E^D) \quad (14.1)$$

Differentiating (14.1) with respect to the tax rate, setting equal to zero, and solving for the tax rate yields the following formula for the *unilaterally optimal domestic tax*:

$$\tau_D = D'(E^D) + s_F^D D'(E^D) \frac{n_F \frac{\partial e_F}{\partial \tau}}{n_D \frac{\partial e_D}{\partial \tau}} - P'_w(Q_w) (n_D q_D - Y_D) \frac{n_D \frac{\partial q_D}{\partial \tau} + n_F \frac{\partial q_F}{\partial \tau}}{n_D \frac{\partial e_D}{\partial \tau}} + P'_w(Q_w) q_D \frac{\frac{\partial q_D}{\partial \tau}}{\frac{\partial e_D}{\partial \tau}} \quad (14.2)$$

The optimal tax consists of four parts: The first term is the marginal damage caused by domestic firms, i.e., the classical *Pigovian tax*. The second term represents the “*leakage effect*,” i.e., how emissions abroad increase as a result of emissions reduction at home. It is given by the domestic damage caused by foreign emissions multiplied by the response of foreign emissions to a domestic tax and divided by the domestic response to the tax. Since $\partial e_D / \partial \tau < 0$ and $\partial q_D / \partial \tau < 0$, the world market price increases inducing foreign firms to increase output and emissions, i.e. $\partial e_F / \partial \tau > 0$. Thus, even without the strategic aspects represented by the third and fourth term, the domestic government should under-internalize the world’s social damage for two reasons. First, it does not take into account the damage to foreign countries ($D'(E^F) s_D^F$ is neglected in 14.2). Second, the tax shifts production and pollution abroad. However, pollution comes back across the border through wind or water. The “*leakage effect*” is always present in cases of transboundary pollution, even if markets are competitive—although the reaction of firms and thus the weight factor $n_F \frac{\partial e_F}{\partial \tau} / n_D \frac{\partial e_D}{\partial \tau}$ depends on the market structure.

On the other hand, the third effect is known as the “*terms of trade-effect*.” Since $P'_w(Q_w)$, this term is positive when the domestic country is an exporter of the polluting good. Finally, the fourth term is known as the “*imperfect competition effect*” and is negative since an increase in the tax reduces domestic output.

Trade, Small Open Economies, and Local Policymaking

Large, closed economies can implement local environmental policies more or less at will, but to understand the importance of trade, it is instructive to look at what pervasive restrictions free trade may impose on a small, open economy. The basic condition in such an economy is that product prices are decided by the world market and cannot be influenced by local policy.

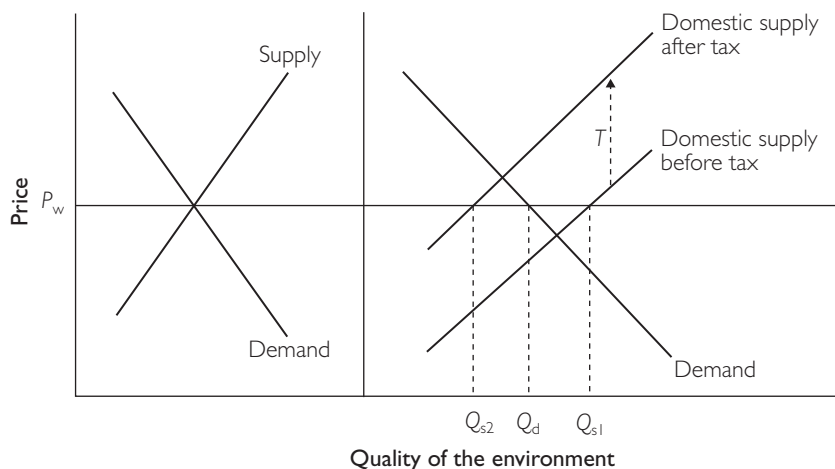


FIGURE 14-1 Effect of Environmental Taxes in a Small, Open Economy: World Market (left) and Domestic Market (right)

Notes: P_w = world market price; Q_s = domestic supply before (1) and after (2) tax; Q_d = domestic demand; T = domestic tax level.

When production per se gives rise to externalities, input or product taxes are needed. However, these instruments are particularly troublesome in small, open economies because they make national production more expensive, leading to reduced exports and increased imports. This effect is illustrated in Figure 14-1, in which the left side shows how the world market sets the equilibrium price P_w . The right side shows the small, open economy with demand Q_d . The local tax has no effect on the world price or domestic demand but depresses the profit margins of local producers so that their supply decreases (from Q_{s1} to Q_{s2}), so the small country ceases to be a net exporter (of $Q_{s1} - Q_d$) and becomes a net importer (of $Q_d - Q_{s2}$). Because the world market price is not affected, total consumption and thus production would be the same; the only effect would be to move production to other countries that do not levy the tax in question.

Relocation results in roughly the same amount of pollution being generated elsewhere (“leakage effect”). If other countries were to use less-efficient or more-polluting production technology and the pollutant in question were truly global, then the net effect of the tax policy could even be to increase global pollution. Short-run costs to the local economy would be lost production, income, and employment, and long-run costs would include exchange rate adaptations and a new equilibrium with, in principle, a lower level of equilibrium salary. Similar effects would result from any policy instrument that raised the aggregate costs of production in the small, open economy. Tradable permits that were auctioned or grandfathered would have the same effect; the effect would be much smaller with output-based permit allocation or a refunded emissions permit scheme (or if a command-and-control mechanism were used; see Chapter 7).

When selecting policy instruments for small, open economies, several special aspects must be considered. First, taxes on production in competitive sectors are not as effective or desirable as in a closed economy. If an instrument is needed for a certain sector (where there is a perceived risk that companies will be hurt by competition from unregulated firms), then instruments such as regulation, output-based permit allocation, tax–subsidy schemes, or refunded emissions payments might be preferable. For protected sectors (e.g., local health care and road building, in which the pressure from foreign competitors is absent or smaller), taxes may still be preferable. An effect that might become problematic with large-scale programs such as carbon trading is that multinational companies trade permits and move abatement costs among subsidiaries in such a way as to move profits from highly taxed countries to tax havens. This effect may be strongest in settings with partial trading regimes (Fischer 2000a, 2000b).

The small, open economy hypothesis is relevant for many developing and formerly planned economies. In fact, policy advice (from the World Bank and others) to such countries is to increase their degree of openness. It is an important step toward economic growth but also results in several restrictions on local policymaking. Many developing countries must address considerable capital flows and major international ownership of important mineral and other natural resources in addition to trade relations. Consequently, external parties have great power, which could create considerable conflict concerning taxes on resource rents or other policies to promote conservation. The private interests of foreign owners must not be perceived as interfering with more general policy advice to countries that are heavily dependent on natural resources for their development. Advising countries with vast untapped natural resources to “open up” their economies is likely to create detrimental overexploitation of these resources if done before they have the capacity to formulate environmental and natural resources policy, enforce regulations, and assign and defend property rights.

In contrast, increased openness is urgently needed for other sectors of developing and formerly planned economies. The continued operation of inefficient plants with distorted relative prices (supported by irrational tariff structures, quotas, or subsidies) has detrimental effects on the local economy and environment. Deregulation is not wrong but should be done carefully, with respect for the inherent risks. In general, micro- and macroeconomic policies must be developed in conjunction with fundamental institutions such as property rights. Structural adjustment is necessary if an economy has been seriously derailed; if interest or exchange rates are out of line with those of other countries, then all other microeconomic policies will be ineffective. Periods of instability or transition can lead to unforeseen effects and heighten the need for good institutions at the micro level in sectors that govern vulnerable natural resources.

In a recent paper, Copeland and Taylor (2009) develop a theory of resource management where the degree to which countries escape the tragedy of the commons depends on three forces: the regulator’s enforcement power, the extent of harvesting capacity, and the ability of the resource to generate competitive

returns without being extinguished. Using this theory they divide resource-rich economies into three groups according to their ability to enforce property rights as world prices vary. They refer to these groups as Hardin, Ostrom, and Clark¹ economies. Hardin economies have limited enforcement power relative to their overcapacity. Therefore, they always exhibit de facto open access (in steady state) and no rents are earned on the resource. Ostrom economies, however, have sufficient enforcement power to generate some rents, but not enough to achieve the first-best. Finally, Clark economies can obtain the first-best at relatively high resource prices, but at low resource prices even a Clark economy exhibits open access or limited management. In such a setting, they analyze the effects of trade liberalization in resource-exporting countries. They show that while trade liberalization leads to resource depletion and real income losses in Hardin economies, the trade-induced increase in relative resource prices can lead to a transition to more effective management in Ostrom or Clark economies, where legal system or monitoring capabilities are better. Since enforcement decisions depend on the terms of trade, an increase in the price of the natural resource can lead to increased enforcement of property rights, which in turn leads to higher steady-state resource stock levels (see also Hotte et al. 2000).

Transboundary Pollution and International Environmental Agreements

Transboundary pollution is one of the most challenging problems in environmental management. It is defined as pollution whose physical origin is within the jurisdiction of one country, but the damage occurs in an area under the jurisdiction of another country. Addressing transnational pollution requires both international and domestic law. At the international level, mechanisms such as treaties, agreements, and international management and governance are required. At the local level, environmental protection can be addressed more effectively and efficiently through the domestic legal system. The challenge in creating and enforcing a transnational pollution control regime lies in harmonizing international and domestic law, which involves three fundamental constraints. First, international environmental agreements (IEAs) have to be acceptable for all potential participants. Second, the parties must agree on the particular design of an IEA by consensus. Third, the treaty must be enforced by the parties themselves. In such a setting, two types of free riding behavior may arise. The first occurs when a country decides not to join an IEA or, if within the agreement, to contribute less to the improvement of environmental quality. This type of free riding, usually known as a *cooperation problem*, is obviously evident and common: in most IEAs the number of signatories fall short of the total number of countries involved in the externality problem. To take one example, in 2001 President George W. Bush announced that the U.S. would withdraw from the Kyoto Protocol because abatement costs from the emissions reduction obligation were expected to exceed the

benefits. The second type of free riding, called the *compliance problem*, occurs when a country does join an IEA but fails to comply with its terms. IEAs will only endure if the incentive to free ride on compliance can be controlled by threats to punish deviations. Therefore, the design of a scheme of sanctions to deter noncompliance is a key issue.

We now discuss the implications of the *cooperation* and *compliance* problems on treaty design in the case of climate change policy. Notice that the problem of designing national and international policies to deal with transboundary pollution problems is akin to the problem in public finance of the division of responsibilities between different levels of government, *fiscal federalism* (Oates and Schwab 1996; Murty 1996). In a federal country, there are governments at many levels (say a central level and provincial governments), and each government levies taxes (probably on the same base) with the central government responsible for distributing part of the collected tax revenue among the provincial governments. Taking fiscal federalism as a point of departure, it is natural to deconstruct the “optimal” tax on emissions into a revenue tax, a tax on local pollution, and an international tax—the proceeds of which could be shared among the countries of the world. This idea is illustrated in Figure 14-2, in which the marginal private benefits (i.e., benefits to

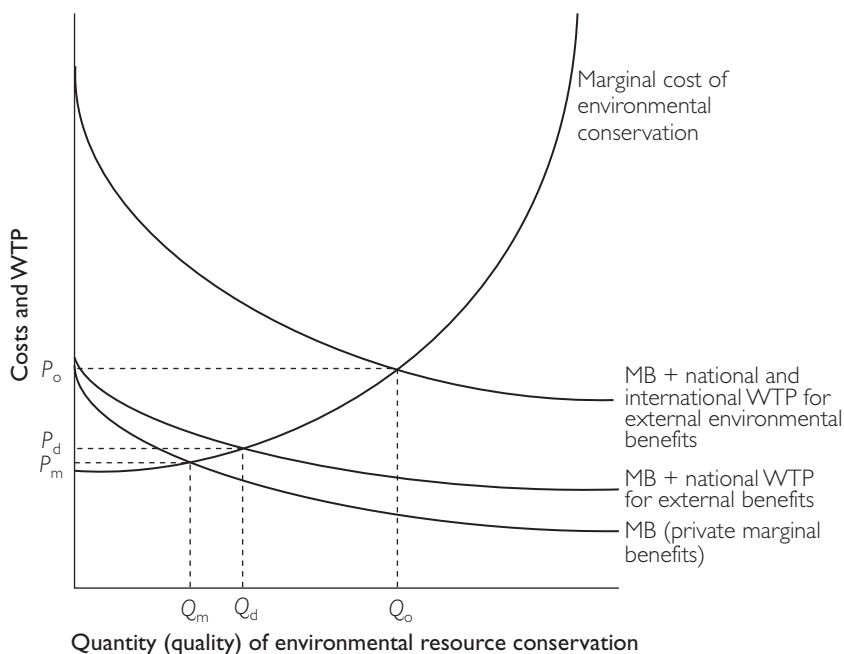


FIGURE 14-2 Private and Social Benefits at Different Levels

Notes: WTP = willingness to pay; P = price; Q = quantity or quality variables; MB = marginal benefits. Subscripts o, d, and m represent global optimum, domestic optimum, and domestic market with no environmental benefits, respectively.

the firm or owner) are small so that the “market” level of conservation is low. Adding local external benefits would, in principle, encourage a higher degree of protection. Equally if not more important in practice is the fact that the likelihood of any conservation may be significantly increased.

Cooperation, Self-Enforcement, and IEAs

Two main reasons seem to explain the difficulties implicit in international agreements that have a large number of signatories. First, even if countries are nearly identical, some countries may prefer to free ride because they can have the environmental benefits, which are not excludable, without paying the costs—a situation reinforced when large economic and environmental asymmetries are spread across world regions. Second, even though global welfare may increase through cooperation, individual countries may be worse off depending on the distribution of the abatement burden. For example, an efficient allocation of the abatement burden requires that developing countries with low marginal abatement cost contribute more than industrialized countries to the reduction of GHG. If differences in abatement burdens are pronounced enough, developing countries may be worse off after joining the agreement if they receive no compensation. Compensation, so important an issue in climate negotiations, can either be overt (monetary side-payments possibly tied to technology acquisition or adaptation) or implicit (for example, in the allocation of rights).

Total global carbon emissions are very approximately 25 gigatons of CO₂. At a hypothetical price of US\$40/ton the total rent would be US\$1 trillion. Many OECD countries argue for an allocation of rights through grandfathering (GF). Arithmetically this is usually presented as (roughly) equal percentage, meaning those countries who polluted the most in the past must do more abatement but still get more generous allocations of permits to emit. The notion of equal percentage reductions (and thus GF) is often taken by its proponents as being a natural or fair principle involving comparable effort. However, other countries make reference to other ethical principles and other allocation mechanisms: low-income countries tend to argue in favor of equal *per capita* allocations. It is striking to realize that the latter principle would give India roughly 16% of total allowances (since India has a sixth of the world population), while GF would only give India around 4% (its current share of world emissions). If each percentage point is potentially worth US\$10 billion/year, one can understand that countries negotiate very carefully. Although it is no wonder climate negotiations take such a long time, given that we are in a hurry to reach a global agreement this is quite bad news.

To more formally describe these issues and analyze solutions proposed in the literature, let us introduce some mathematical notation. Assume that the negotiations take place among n countries, $i = 1, 2, \dots, n$, and that the welfare of country i is given by the function $\pi_i = \beta_i(e_i) - \varphi_i\left(\sum_{j=1}^n s_j^i e_j\right)$.

Notice that country i benefits from its own emissions e_i —which can be viewed as an input in the production and consumption of goods; but it also suffers damages from its own emissions and the emissions released by the other countries $[e_j, j \neq i]$. As before, the coefficients $s_j^i \in (0,1)$ are pollutant transfer coefficients. For global pollutants, like CFCs and GHG, these coefficients are close to one since emissions disperse uniformly in the atmosphere. However, in the case of other transboundary pollution problems—as for example, acidic precipitation—“upstream” and “downwind” pollution do not have the same impact.

Since the regeneration capacity of most environmental systems is limited, environmental damages increase more than proportionally with increasing depositions. Hence, we assume damages increase in depositions $\varphi'_i > 0$ at an increasing rate $\varphi''_i \geq 0$. Finally, it is usually assumed that emission benefits increase $\beta'_i > 0$ at a decreasing rate $\beta''_i < 0$ due to diminishing returns.

If each of the n countries behaves noncooperatively—that is, pursues its own interest, maximizing welfare function with regard to its own level of emissions and taking emissions from the other countries as given—the simultaneous solution of the n first order conditions of the maximization problems delivers the noncooperative Nash equilibrium vector $e^{NC} = (e_1^{NC}, \dots, e_i^{NC}, \dots, e_n^{NC})$. In contrast, if governments behave fully cooperatively pursuing the common interest, they would maximize the aggregate payoff over all countries $\sum_{i=1}^n \pi_i$. The simultaneous solution of the n first order conditions of the cooperative problem this time would deliver the socially optimal emissions vector $e^O = (e_1^O, \dots, e_i^O, \dots, e_n^O)$.

Countries facing an international environmental agreement play a two-stage game. In the first stage—usually known as the *coalition stage*—they decide noncooperatively whether or not to sign the agreement. In the second stage, they play a noncooperative Nash equilibrium game, where each country which signed the agreement plays as a single player—although a given burden-sharing rule is pre-specified as a part of the agreement.

We can assume that all countries decide simultaneously and that only one coalition can be formed, which implies that those countries that defect from the agreement play as singletons. In this case we can say that the coalition is profitable when each country gains from the agreement. Formally, $\pi_i(e_1^O, \dots, e_i^O, \dots, e_n^O) > \pi_i(e_1^{NC}, \dots, e_i^{NC}, \dots, e_n^{NC}) \forall i$. On the other hand, the coalition is stable (self enforcing) if there is no incentive to free ride, i.e. $\pi_i(e_1^O, \dots, e_i^O, \dots, e_n^O) > \pi_i(e_1^O, \dots, e_i^{NC}, \dots, e_n^O) \forall i$.

Most articles in the theoretical literature on environmental cooperation and conflict have found that the presence of asymmetries across countries and the incentive to free ride makes the existence of self-enforcing agreements quite unlikely. If the agreements exist, they are signed by a limited number of countries, while the number of signatories is larger when the gains from the agreement—that is, the difference between the benefits of cooperative and noncooperative behavior—are small. (See for example, Hoel 1991; Barrett

1997b; Carraro and Siniscalco 1998.) Kolstad (2010b) examines how heterogeneity among countries affects the development of IEAs. The question is whether heterogeneity (as opposed to homogeneity) retards or enhances the formation of IEAs, leads to higher or lower levels of aggregate welfare, or to more or less abatement. As expected, he concludes that heterogeneity of countries may increase the size of IEAs, but it weakens them in terms of the aggregate welfare of all participants.

These results suggest that the attempt to negotiate on emission reduction is unlikely to be successful unless more complex strategies are implemented. Four main sets of instruments have been proposed to increase the number of signatories of environmental agreements. These instruments are *minimum participation*, *transfers* among signatories, the *linkage* of environmental negotiations to other economic issues, and *economic sanctions*.

Many IEAs contain a *minimum participation* clause which stipulates that the treaty does not enter into force unless it is ratified by at least a critical number of countries. For example, the Kyoto Protocol became binding only after being ratified by not less than 55 countries. Some authors believe that this strategy could enforce cooperation, provided that participation is profitable. If a country fails to sign the agreement, all countries revert to the noncooperative equilibrium which makes them worse off. However, although it can serve as an important coordination device, its selection is at risk due to the existence of an inferior equilibrium. As long as the ratification falls short of the minimum participation level, the noncooperative equilibrium will be reached and hence no country loses by ratifying the agreement. On the other hand, if the minimum participation clause is matched, the cooperative equilibrium is sustained and so the clause makes ratification a weakly dominant strategy (see Barrett 1998 and Wagner 2001).

Transfers are often proposed to compensate those countries that lose by signing the agreement. A well designed IEA can guarantee that no country refuses to sign the agreement for fear of unprofitability. Formally, this implies that $\pi_i(e_1^O, \dots, e_i^O, \dots, e_n^O) + T_i > \pi_i(e_1^{NC}, \dots, e_i^{NC}, \dots, e_n^{NC}) \forall i$, where T_i denotes the transfer given to country i . Transfers may also offset incentives to free ride. However, to sustain broader coalitions by means of transfers, a minimum number of countries must commit to finance cooperation (Carraro and Siniscalco 1993). How many countries can be induced to sign the agreement using this strategy? The answer depends on the impact of foreign emissions on other countries' welfare. If the impact is high, then the best noncooperative response of a country to an expansion of foreign emission is an emission reduction. On the contrary, if foreign countries agree on an emission reduction and some countries do not sign the agreement, the nonsignatories' best response to the agreement is an increase in their own emissions, which offsets the effort of the cooperating countries. The pattern of interdependence among countries is crucial for determining the sustainability of cooperation; a larger degree of interdependence increases the extent of free riding and reduces the expected

gains of cooperation. If the degree of interdependence is small, the emissions reduction by signatories is not offset by free riders, who simply enjoy a better environment but do not directly damage countries which cooperate. Then, stable coalitions are more likely to occur.

Transfers are in practice an important feature of international agreements. Take, for example, the Montreal Protocol. Although side payments were not included in the original agreement in 1987, the treaty was amended in 1990 to compensate developing countries for the “incremental costs” of their participation (Barrett 2001). For that purpose, the Multilateral Fund was established in 1991 to assist developing country parties to the Montreal Protocol—defined as those whose annual per capita consumption and production of ozone-depleting substances was less than 0.3 kg—in meeting compliance with the control measures of the Protocol. Soon after the amendment was negotiated, the number of developing countries in the treaty shot up. As of July 2009, contributions made to the Multilateral Fund by some 49 industrialized countries totaled over US\$2.5 billion. The Fund Executive Committee which manages the Fund has approved the expenditure of over US\$2.3 billion to support over 6,000 projects and activities in 148 countries. Implementation of these projects is expected to result in the phase-out of the consumption of more than 258,574 tons of ozone-depleting substances and the production of 195,013 tons.

Carraro et al. (2006) highlight the relevance of transfer schemes to climate change policy. They use an integrated assessment model of climate change policy to identify the agreement that would emerge at the equilibrium. They find that transfers paid both by signatories and by nonsignatories may lead to far better results in terms of inducing almost all countries to sign a climate treaty. The latter result has important implications for post-Kyoto climate policy negotiations because it highlights the important role that outsiders—like the U.S.—can play. Even if it does not join the Kyoto Protocol, the U.S. may be involved in future protocols via financial mechanisms through which they could sponsor developing countries to assume quantitative emission ceilings and to become full members of the Protocol. These transfers might be in the interest of the U.S. if it would increase global greenhouse gas emission control and reduce the cost of taking domestic emission control measures.

As for transfers, the *linkage* of environmental negotiations to other economic issues may also ensure that all parties gain by participating in an agreement and discourage free riders. The intuition behind this strategy is that IEAs are *club goods agreements* that suffer from free riding since the gains from cooperation are nonexclusive to signatories. Instead, club goods agreements enjoy a higher participation since the gains from cooperation are exclusive to signatories. Consequently, linking a public good agreement may increase IEA participation and success. Examples of club goods agreements that have been proposed in the literature include trade agreements and agreements on R&D. Under a trade agreement, for example, firms in member countries have a competitive advantage over outsiders; under an R&D agreement, however,

they share the costs of R&D, implying that those firms reduce their marginal and average production costs compared to firms in nonmember countries. Although in theory *linkage* may increase participation, in practice membership may be mixed. It may be difficult to exclude countries from a trade agreement or defense pact if they do not join an IEA. Besides, the positive effects may be overestimated if in the context of many countries the linkage issue involves time-consuming negotiations or if it lowers participation from the perspective of the club goods agreement (see Finus 2003).

The number of signatories could also be increased if nonsignatories are threatened to be punished through *reciprocal measures*, *financial penalties*, and *trade sanctions*. Indeed, if the game outlined above is repeated ad infinitum, strategies based on reciprocity could be exploited to enforce cooperation. To see this, suppose that a country defects once and then reverts to cooperation, which gives it the maximum gain from cheating for one period. To deter free riding, the compliant countries must punish the defection by lowering their own levels of abatement. In doing so, these countries realize lower benefits than under full cooperation. Unfortunately, this strategy might be difficult to establish in the context of climate change mitigation since it is unlikely that countries will sustain self-damaging levels of emissions (a reduction in mitigation) in order to punish infractions. Indeed, this strategy is only rational if the self-harming punishment induces the deviant country to revert to cooperation; that is, if the long-run gains from full cooperation outweigh the loss from punishing the deviation.

Enforcement schemes that impose *self-punishment* have also turned out to have some other problems. For example, discussions of sanctions for noncompliance within the Kyoto Protocol focused on how noncomplying countries might be required to make up for excess of emissions in subsequent compliance periods. At the extended Sixth Conference of the Parties in Bonn in 2001, countries agreed to apply a noncompliance penalty ratio of 1.3, meaning that if an Annex I party were to emit x tons more than allowed in the first compliance period (2008–2012), then the party's emission cap for the next compliance period would be reduced by $1.3x$ tons – x tons to offset the excess plus an additional penalty for noncompliance equal to $0.3x$. Although this proposal seems to be attractive since it does not require sacrifices by compliant countries, it is precisely for that reason it can be expected to fail to affect behavior. First, because the punishment could be forever delayed. Second, the magnitude of the punishment depends on the agreed penalty rate, but also on future emissions; if a country fails to comply in the first compliance period, it has the incentive to negotiate a relatively lax emission cap for the next compliance period. Finally, the proposal seems to ignore that the easiest way of avoiding future penalties is to withdraw from the agreement at a subsequent date.

Financial penalties are frequently used to punish noncompliance with domestic regulations. However, in the international context, no international authority has the ability to impose credible penalties of sufficient magnitude.

A third type of negative incentive for compliance is *trade restrictions*—for example, import tariffs. They can promote greater participation in the treaty by shifting consumption toward the cooperative country since domestic producers in treaty countries would have a price advantage. Although the World Trade Organization (WTO) has evolved in the direction of supporting environmental protection—even at the expense of some disruption in trade—applications of trade sanctions would pose a massive challenge to the trading system and may be subject of manipulation. The mere suspicion of such behavior could very well make future trade negotiations more acrimonious, and this would itself carry big costs.

In general, a combination of carrots and sticks may be more effective while also enhancing the perceived fairness of an international treaty. This was the case with the Montreal Protocol, where developing countries that stood to gain the least from ozone protection were compensated for participating in the agreement, making the threat to impose trade restrictions in products containing chlorofluorocarbons (CFCs) appear as fair. Compensation was instrumental in sustaining cooperation. In practice, trade has not actually been restricted, but the belief that trade would be restricted if countries failed to participate had the effect of promoting participation.

Uncertainty and risk aversion also might affect the emergence of cooperation (see Box 10-1). Boucher and Bramoullé (2007) introduce uncertainty and risk aversion in a model of treaty formation. They show that uncertainty about treaty payoffs reduces the signatories' efforts to reduce pollution but increases the critical mass of countries at which cooperation becomes sustainable; this participation increase can yield an increase in welfare. Nevertheless, this positive effect of uncertainty might be mitigated by the existence of multiple equilibria: Countries can also be trapped in a low participation equilibrium, where participation is not affected by risk and welfare is much lower.

The Economics of Controlling Stock Pollutants and Climate Change Policy

A complication in the policy response to climate change is the time delay in the reaction of climate to increasing atmospheric GHG concentrations. A part of this is the inevitably long time it takes to heat the ocean's mass of water. Estimates of the delay to equilibrium range up to around 100 years (Nordhaus 1991). Because GHG emissions are persistent, the levels of pollution in neighboring periods are causally related. If we apply optimal control theory to develop a strategy to control GHG, the optimal strategy will require that any time the marginal costs of abatement should be equated with the present value of the marginal damage of timely unabated emissions. The optimal policy strategy would call therefore for increasingly tight emission levels over time as the pollutant stock accumulates. To clarify this result, let us introduce some notation.

Let S_t denote the cumulative stock of the pollutant, E_t the flow of emissions, and A_t the number of abated pollution units at time t . The pollutants accumulate according to the following equation of motion:

$$\dot{S}_t = \frac{dS_t}{dt} = E_t - A_t - gS_t. \quad (14.3)$$

where g represents a constant decay rate of pollution. Define $D(S_t)$ as the value of the social damage associated with S_t , $C(A_t)$ denotes the abatement cost function, and r represents the discount rate. If the regulator wants to implement an abatement strategy that minimizes the present value of all emission-related costs of the pollutant, the optimization problem can be represented as:

$$\min_{A_t} \int_0^{\infty} [D(S_t) + C(A_t)] e^{-rt} dt. \quad (14.4)$$

This problem is subjected to the equation of motion of the accumulating stocks. As described in detail in Chapter 22, Hamiltonian functions are the most common approach to solve dynamic problems that involve continuous time. In this case, the Hamiltonian is then given by:

$$H = D(S_t) + C(A_t) + \mu_t [E_t - A_t - gS_t] \quad (14.5)$$

where $\mu_t = e^{-rt}\lambda_t$ represents the social cost of an increment on the stock of the pollutant at time t in monetary units of time t . λ_t represents the cost of an increment on the stock of the pollutant at time t in monetary units of time 0.

Since A_t is the variable of control and S_t is the state variable, the first-order conditions of this problem are:

$$H_A = 0 \quad (14.6)$$

$$H_S = -\dot{\mu}_t + r\mu_t \quad (14.7)$$

So in our case we get:

$$C'(A_t) = \mu_t \quad (14.8)$$

$$D'(S_t) = -\dot{\mu}_t + [r + g]\mu_t \quad (14.9)$$

Finally, to obtain an expression for $\dot{\mu}_t$, we differentiate equation (14.8) with respect to time, such that $C''(A_t) \frac{dA_t}{dt} = \dot{\mu}_t$. Then, the optimal path of abatement can be represented as:

$$C'(A_t) = \frac{D'(S_t) + C''(A_t) \frac{dA_t}{dt}}{r + g} \quad (14.10)$$

Thus, along the optimal path of abatement, the marginal cost of abatement should be equated with “the present value of the marginal damage caused by emissions plus the present value of any change in the marginal abatement costs due to unabated emission units.” Notice also that the rate of discount rate reduces the value of the damage of future concentrations caused by current emissions. Therefore, a higher value of r reduces the desired level of current abatement expenditure (see Falk and Mendelsohn 1993).

In this setting, the social damage function is time invariant. However, in practice, several factors might affect the damage function through time—for example, population growth, economic growth, and technical progress in adaptation. Population growth could cause the damage per ton to increase over time, while an increase in income is likely to affect the willingness to pay for environmental improvements. On the other hand, technical change in adaptation is likely to reduce damages over time. Abatement costs also can vary due to factors such as technical change and the abundance of nonrenewable resources used in the abatement process.

Hoel and Karp (2002) compare the effects of taxes and quotas for an environmental problem in which the environmental damage depends on the stock of pollution and the regulator and polluter have asymmetric information about abatement costs. They show that the intuition provided by Weitzman’s model continues to hold. However, when environmental damages depend on pollution stocks, the ranking of the two policies also depends on the discount and stock decay rates. Higher discount or higher decay rates increase the importance of current flows relative to future stock effects. In this setting, the main advantage of the quota is that it makes it possible to control exactly the evolution of the stock. Since a higher discount rate and a higher decay rate both decrease the importance of future stock effects, they also decrease the value to the regulator of exact control of the evolution of the stock. Consequently, higher discount and decay rates make it more likely that a tax is preferred. They also show that the ranking of the two policies also depends on the length of the period for which agents keep their decision variables unchanged; the regulator always prefers taxes if the length of this period is very small. However, the difference between the payoffs vanishes as the length of the period becomes small.

Equity, Discounting, and Climate Change Policy

The fairness of climate change policy is at the core of the debate. The two main aspects involve *international equity* and *intergenerational equity*.

International Equity

International equity plays a key role in dealing with climate change impacts, adaptation, and climate change mitigation. Clearly, countries differ in their

vulnerability to climate change, their contributions to global emissions, and their ability to bear the costs of mitigating GHG emissions. On the side of the impacts, the IPCC Second Assessment Report made clear that the effects of climate change will vary greatly among countries; doubling concentrations of GHG emissions in the atmosphere could cost on the order of 1.0%–1.5% of the gross domestic product (GDP) for developed countries and 2.0%–9.0% of GDP for developing countries. In hotter climates and developing countries whose economies are more dependent on natural resources, the impacts are widely expected to be even worse. These regions are currently contributing the least to the problem, but they are likely to be the most vulnerable—creating a very important equity tension.

Mitigation has been the focus of debate in past decades due to the urgency of promoting international action to reduce the causes of human induced climate change. Differences in perceptions about what constitutes an equitable distribution of mitigation efforts complicate any agreement. Several developing countries continue to press the claim that the Annex I countries should do even more to reduce GHG emissions given their historical responsibility. Developed countries accept the concept of differentiated responsibilities, but they want greater assurance on when and how developing countries would start assuming greater responsibilities, since they are likely to become primary industrial emitters in the future.

This ongoing debate illustrates how differently countries interpret what constitutes fairness or equity in mitigating the risks of global climate change, as was illustrated earlier by our discussion on grandfathering versus equal per capita allocation. Current emission intensities vary dramatically between rich and poor regions (see Table 14.1), although developing countries have

TABLE 14-1 International Equity and Climate Change

<i>Country</i>	<i>Annual CO₂ emissions 2006 (in thousands of metric tons)</i>	<i>Annual CO₂ emissions per capita 2006 (metric tons)</i>	<i>Income per capita 2008 (US\$)</i>
China	6,103,493	4.65	5,962
United States	5,752,289	19	46,716
EU	3,914,359	8.65	36,901
Russia	1,564,669	11	16,139
Latin America	1,513,168	2.33	7,722
India	1,510,351	1.36	2,972
Japan	1,293,409	10	34,099
Africa	1,067,933	1.19	2,671
Australia	372,013	18	35,677

Source: World Bank Statistics and United Nations Statistics Division, Millennium Development Goals indicators: Carbon dioxide emissions (CO₂), thousand metric tons of CO₂.

been increasing their emissions over the last decades as a consequence of economic growth. Indeed, since 2006, China has led the world in terms of total CO₂ emission (which is mainly explained by its large population), followed closely by the United States. These two countries account for about a half of the world emissions, but they represent very different perspectives on what a fair allocation would be.

Given the current distribution of emissions across countries, the choice of an equity principle to decide on the burden sharing of climate change is not irrelevant. The *egalitarian* principle is popular with many developing countries. When countries are assumed to reduce emissions in proportion to equal per capita emissions, most abatement responsibilities shift to developed countries where per capita emissions are much higher than in developing countries. The *ability to pay* is another principle favored by developing countries; it allocates the burden across nations relative to economic circumstances, so net cost proportions are inversely correlated with per capita GDP. Wealthy nations would pay most of the cost, although developing nations would have to pay more as they become wealthier over time. In the same line, the *polluter pays* principle proposes to share abatement costs across countries in proportion to emission levels, implying a reallocation of burdens over time as developing countries become larger emitters. Contrarily, the *sovereignty* principle implies symmetrical costs burdens: emissions reductions across countries maintain current relative emission levels among them (as with grandfathering). Finally, the *Kantian allocation rule* proposes that each country chooses an abatement level at least as large as the uniform abatement level it would like all countries to undertake, ensuring that each country's true preferences for abatement are revealed. Each country takes on a relative cost burden equal to what it expects of others (see Cazorla and Toman 2001 for further discussion). Based on a worldwide survey of people involved in international climate policy (such as negotiators or observers), Lange et. al (2007) analyze the importance of equity empirically. They find that the polluter pays rule and the egalitarian principle are the most widely accepted equity principles. As expected, persons from richer countries are less in favor of incorporating the polluter pays and ability to pay principles in future international climate agreements, while the egalitarian rule is more strongly supported by individuals from countries with lower per capita GDP at present.

All of these principles have their own problems, and it is likely that none of them can be expected to govern resolution of the international equity issue. However, approaches that involve adjusting responsibilities over time on the basis of more than one criterion may offer more promise for successful negotiation over the long run.

So far, most discussions on equity focus on mitigation of climate change by emissions reductions, ignoring the newly understood need for adaptation to climate change. Long-standing unease about adaptation may come from fears that acknowledging the necessity for adaptation could distract international

efforts to mitigate climate change. However, we must admit that some of us will have no option but to adapt to climate changes, sometimes to which some of us have not contributed. Adaptive responses might involve changes in the behavior of affected households (such as switching crops); investments in transport systems, water storage capacity, and flood protection; improved buildings; and most dire of all, in some cases migration.

Adaptation presents several dilemmas to the global community. To what extent should developed countries support adaptation by developing countries? What should be the distribution of the burden among developed countries? Questions like these represent big challenges for the management of natural resources and policymaking. It is also problematic that we know so little about the size of the adaptation costs. Focusing on how policies can support the adaptive capacity and resilience of vulnerable communities will be a crucial task in the coming years.

Discounting, Intergenerational Equity and the "Even Sterner" Review

How much climate change mitigation should we undertake today? This is a hard question to answer in part because it depends on our view of the distribution of resources across generations. Greenhouse gas emission reduction over the near term would mitigate future damages, but even large reductions will do little to alter the present climate and the present rate of change in climate impacts. The costs of emission abatement must therefore be justified by the benefits of avoided impacts well into the future. The rate of discount is at the center of this discussion. High rates may leave future generations with unacceptable climate damages.

Why should society discount future avoided impacts? At least two reasons make it reasonable to do so. First, an additional unit of consumption tomorrow would be of less value than an additional unit of consumption today if society is impatient to enjoy that additional unit now. Second, concerns about intergenerational equity and justice provide an argument for consumption to be evenly spread across generations; if future generations are likely to be richer than us, there is a case for valuing an extra unit of their consumption less than an extra unit of our consumption. To understand the logic behind discounting, we can make use of the Ramsey model of economic growth. The Ramsey model derives the discount rate (r) from three parameters: ρ is the "pure time preference" and accounts for the weight placed on a unit of utility in the future compared with an equal unit of utility experience today; η accounts for social value judgments about the distribution of income among generations, i.e., the weight placed on the consumption of (current) poor compared to the consumption of the (future) rich generations. Finally, g accounts for the growth rate of consumption per capita.

$$r = \rho + \eta g \quad (14.11)$$

Notice that r increases with ρ and g , but it increases with η if and only if $g > 0$. This means that just as growing consumption provides a reason why discount rates should be positive, declining consumption would be a reason why they could be negative. This issue is of importance when it comes to discussing the optimal timing of climate policy and the effects on people in the tropics, who are currently very poor and very likely to suffer greatly from climate change under business as usual (Dasgupta 2008).

There is a great debate about what reasonable values for ρ , η , and g are. The *Stern Review*, the first official economic report to give climate change a prominent place among global problems, adopted the values $\rho = 0.1$ and $\eta = 1$; when combined with a rate of consumption growth of 1.3 percent, he arrives at a very low discount rate of 1.4%. This rate has been highly criticized on several grounds. First, Nordhaus (2007) and Dasgupta (2007) pointed out that these numbers do not match the observed market rate of interest and savings rates. Second, several authors have argued that η should be higher, condoning a range of 2 to 4 based upon revealed preferences over gambles (see for example, Weitzman 2007). Finally, some are concerned about the extent to which economic growth can go on—and about the material and ecological sustainability of this economic growth. In an earlier analysis, Azar and Sterner (1996) raised the idea that growth cannot “go on forever.” They assumed that growth would continue only until we became ten times richer and that then income would level off. This simple assumption leads to declining discount rates, increasing the present value of avoided damages in the future very significantly. Biologists and engineers find it obvious that we cannot have growth in a finite physical system. On the other hand, economists are loath to think of limits to growth. If one does not accept such limits one clearly has to envisage that growth will be restricted to sectors that are not very intense in their use of physical material.

The insight that future growth means that some but not all sectors grow prompted Hoel and Sterner (2007) to focus on the consequences of uneven growth for discounting: they show how to integrate the effects of the changing composition of the economy and show that when discounting goods in a sector that grows slowly there is an effect of changing relative prices, which also depends on the growth rates. For an environmental resource that will become scarce in the future, the relative price will typically rise. This effect may counteract the effect of discounting, and if an item's rate of price increase were fast enough, the effect might even be reversed, increasing the weight of future environmental damages in economic calculations and the economic benefits of early mitigation measures. In “An Even Sterner Review,” Sterner and Persson (2008) use this principle to comment on the debate between Stern and Nordhaus concerning the discount rate. Nordhaus has asserted that the *Stern Review* results only depend on using an artificially low discount rate. However, Sterner and Persson (2008) reproduce “Stern-type” results even when using discount parameters as high as Nordhaus's—by assuming there

is one sector of the economy for ecosystem services that does not grow and which is hard to substitute for. This sector experiences rising relative prices, making future damages to it expensive.

At the core of this discussion is the possibility of substituting deteriorated environmental quality by other goods. If the substitutability is limited, the environmental deterioration effect dominates the economic growth effect, and the discount rate should be small or negative, thereby inducing us to preserve the environment (see also Gollier 2010).

Analysts who see the need for urgent action to address the threat of climate change frequently invoke the tremendous uncertainty about its impact. An important feature of interest rates under uncertainty is that they do not aggregate arithmetically into a simple certainty-equivalent interest rate (Gollier 2007; Weitzman 2007). Instead, if there is a subjective probability p_i that the discount rate r_i is the correct rate to use, then the effective discount rate for time t is:

$$r(t) = -\frac{\ln \sum p_i e^{-r_i t}}{t} \quad (14.12)$$

This rate decreases monotonically over time from the expected interest rate $r(0) = \sum p_i r_i$ to an asymptotic limit of $r(\infty) = \min_i \{r_i\}$. According to Weitzman (2007), the paradox is that the Stern's estimates may end up being more right than wrong when fully accounting for the uncertainty of the discount rate itself. The very same force of compound interest that makes costs and benefits a century from now relatively insignificant also forces us to recognize the logic that over such long periods we should be using interest rates at the lower end of the spectrum of possible values (Weitzman 2007).

Pesaran et al. (2007) and Weitzman (2008) have shown that the probability distributions over the uncertain $r(t)$ can plausibly have a thick lower tail, implying that a long run of low realizations of $r(t)$ would not be unlikely. When these heavy tails are combined with very unsure high-temperature damages, this aspect can dominate the discounting aspect in calculations of expected present discounted utility and place severe limitations on the reliability of policy advice coming from a standard cost-benefit analysis of climate change. Weitzman (2008) points out that the conventional economic advice of spending modestly on abatement now but gradually ramping up expenditures over time is an extreme lower bound on what is reasonable rather than a best estimate of what is reasonable.

A new line of input into this debate comes from the literature on endogenous technical progress. Acemoglu (2010) for instance characterizes the debate by saying there are two positions:

- 1 The Nordhaus approach leading to the conclusion that the intervention needed is limited and should be introduced gradually—for instance with low and slowly rising taxes.

- 2 The Stern/Al Gore approach that we need a large intervention immediately and permanently.

As already pointed out the choice appears to hinge on the discount rate. However, Acemoglu shows that equally crucial are assumptions on technical change. They assume cumulative technical progress—which the authors refer to as “researchers working on the “shoulders of giants”—*in their sector*. Thus advances in dirty (clean) inputs make their respective future use more profitable. In this setting the policy interventions can *redirect technical change* towards clean technologies. The conclusions depend of course on some crucial parameters but paint quite an optimistic picture: Some of the key results are that two instruments (both carbon tax and technology subsidy) are needed, and that the cost of delaying action today can be very high. Under some set of plausible parameter values the carbon tax might only need to be temporary. The important message is to start *immediately* to stimulate technical progress. If this is truly successful, its impetus will strongly reduce the need for a carbon signal.

Another important result is that if the elasticity of substitution is high enough and technical progress sufficiently successful the discount rates do ultimately not matter—the effect is dominated by cumulative technical progress.

Supplemental Reading

Azar and Sterner 1996	Falk and Mendelsohn 1993
Barrett 1994, 1997a and b, 2000, 2001	Gollier 2010
Carraro and Siniscalco 1998	Hoel 1991
Copeland and Taylor 2009	Nordhaus 2007
Dasgupta 2007, 2008	Sterner and Persson 2008
	Weitzman 2007, 2008

Notes

- 1 They named this third class of economies after the mathematician Colin Clark, whose book *Mathematical Bioeconomics* has played a major role in the teaching and study of resource economics.

Design of Policy Instruments

In real policymaking, all the aspects discussed in Part III must be blended together and must interact with the intricacies of other policymaking. To facilitate a dialogue with the policymaker, it is desirable to summarize the modeling results of instrument design under various conditions in a policy selection matrix (PSM).

Such an exercise may be possible for a concrete issue, for which the number of choices can be limited, but at the general level of the present discussion, it is impossible to do justice to all the complexities in only one table. However, for the sake of illustration, Table 15-1 presents the outline of such a PSM.

Environmental Policy Selection Matrix

Row 1 of Table 15-1 shows the preferred policies for static efficiency (i.e., cost efficiency in a static allocative sense). It is the most classical economic argument explained in many textbooks. The basic concept is that if the differences in marginal costs of abatement (but not in damages, where perfect mixing is assumed) are large, then considerable savings are possible by allowing the agents that have the lowest costs of abatement to do most of the abatement. The regulator usually is assumed to have imperfect knowledge about costs and may thus choose uniform reductions when using command-and-control instruments. Market-based instruments (MBIs) are preferred because they tend to equalize marginal costs and thus allocative efficiency in the use of resources for cleanup. Row 2 is a closely related situation in which the marginal cost of abatement is steep and the benefits of abatement are flat; in this case, price-type instruments are generally the best choice.

In contrast, Row 3 reflects a situation in which the costs of environmental damages vary considerably, giving individual licensing, detailed time-of-day or zoning regulations a clear advantage. The alternatives (e.g., ambient permits or differentiated taxes) may be too complex. When it is necessary to secure a definite level of goal fulfillment, as in Row 4, most MBIs rely on an estimate of responsiveness (i.e., an elasticity) to achieve the correct response level. If

the country is bound by treaties or if there are considerable thresholds in the damage function (or if it is steep) and it is absolutely necessary to meet a goal exactly, then searching for the appropriate tax or charge level would be risky. Physical regulation is preferred in this case, but regulating one source at a time can cause problems for aggregate pollution and thus for ambient levels. The best instrument may thus be the tradable permit that combines the features of a market mechanism with a design that ensures aggregate goal fulfillment.

Row 5 shows how the instruments adapt to changes such as inflation; complicated changes such as economic growth and entry (presented in Chapter 9) are not discussed here. The basic principle is that quantity-based instruments give a constant emissions level that is desirable (e.g., given inflation and maybe entry). Price-based instruments give a constant price signal that is undesirable in the face of inflation; inflation will weaken the policy, which normally is undesirable. (One exception could perhaps be subsidy instruments, because permanent subsidies can have many negative effects and thus it might be good if the subsidy were gradually reduced by inflation.)

Dynamic efficiency is a measure of how well a policy instrument encourages the efficient use of resources—including environmental resources—when faced with rapid technical change (Row 6). If the marginal costs of abatement decrease by 50% per year, then this year's socially efficient level of pollution will be excessive a few years hence. Under a command-and-control policy, the regulator will be forced to either update permits often or accept that the wrong levels prevail for a long time. In principle, similar problems could occur with the appropriate level of a tax or charge, but when the marginal damage of pollution is relatively constant, these instruments are superior because they continue to provide roughly accurate incentives for abatement without adjustment. In this situation, tradable emissions permits would not adjust automatically; a more elaborate design would be needed, with frequent revisions in the number of permits.

It is difficult to make taxes sufficiently detailed for complex environmental conditions (e.g., some industrial chemicals and agricultural pesticides), as indicated in Row 7. Many chemicals are potentially hazardous, and although product taxes could be used in some contexts, creating a sufficiently detailed set of taxes that could discriminate between substances would be a difficult task. In choosing between two products or two processes, the amount of information available is so vast that it must be distilled to make the costs of information transfer and assimilation manageable. If there were individual charges on each chemical, then the market mechanism would perform this interpretative function; however, this approach is hardly practicable. Thus, in this case, other instruments have an advantage. The best policy instruments for some kinds of complex industrial processes may be traditional licensing procedures, which allow an authority to weigh many factors and specify various rules, or voluntary agreements. At the other end of the scale is information disclosure. In some cases, green labeling appears to work efficiently in condensing

TABLE 15-1 Example of a Policy Selection Matrix

Row no.	Criteria and conditions	Taxes and charges	Two-tiered instruments	Subsidy	Tradable emissions permits	CAC instruments	Information disclosure and other
1	Static efficiency	<i>Best:</i> With heterogeneous abatement costs, MBIs save on costs.				Costs are high, especially with mandated technology.	
2	Marginal costs steep, benefits flat	<i>Best:</i> With flat benefit curves or steep abatement curves, price-type policies are preferred.					
3	Efficiency with heterogeneity in damage costs	Tax variation is perceived as unfair and impractical. Modern technology may increase feasibility.			Ambient permits are difficult.	<i>Best:</i> Licensing, zoning, and regulation are preferred.	Information provision and liability legislation are necessary prerequisites or substitutes.
4	Marginal costs flat, benefits steep	Exact goal fulfillment is difficult because MBIs rely on demand estimates. Low fees lead to excessive pollution.			<i>Best:</i> Aggregate emissions control is achieved.	Individual but not aggregate emissions are controlled.	
5	Inflation	Price-based instruments are sensitive.			Unaffected by inflation.		
6	Dynamic efficiency	MBIs imply cost savings and intertemporal efficiency. Subsidies distort entry and exit conditions. Attention to details of permit allocation and refunding is necessary. Quantity instruments are less satisfactory with fast technological progress.				CAC licenses offer no incentive.	
7	Complexity	Technical or ecological complexities put MBIs at a disadvantage.				License, liability, VAs, or information disclosure may be best.	
8	Distribution and political issues	Taxes are unpopular with polluters; charges may be preferable.	Flexibility is an advantage.	Most popular with polluters.	Flexibility is related to permit allocation mechanism.	Polluters like because they pay only for abatement.	Information disclosure is a first step. Liability may be important.

TABLE 15-1 (Cont.)

Row no.	Criteria and conditions	Taxes and charges	Two-tiered instruments	Subsidy	Tradable emissions permits	CAC instruments	Information disclosure and other
9	Asymmetric information and risk, nonpoint-source pollution	Hard to use without good monitoring. Ambient taxes are difficult.	Best: Self-revealing contracts are preferred.	Ultimate policy if no polluter can be found.	Hard to use without good monitoring.	Mandatory technology is easy to inspect.	Labels, liability, and CPR management are good examples.
10	Small number of polluters	Unsuitable for reasons of administrative efficiency and thin markets.				May be best for small numbers.	
11	Rent-seeking	Engenders opposition.	Has good potential.	<i>Bad:</i> Leads to rent-seeking.	May be used as barrier to entry.	Individual negotiation has risks.	Subsidy removal is needed.
12	General equilibrium	Optimal due to output and revenue-recycling effects.	Has no output effect.	Perverse output effects are costly for the state.	Output effect but no revenue unless permits are auctioned.	Normally not applicable.	
13	Developing economy	Inflation and corruption complicate price-type policies. Environmental funds are attractive. Political aspects of monitoring and enforcement need attention.			Promising, but legal issues must be resolved.	Information disclosure and regulation are natural starting points. CPR management may be useful. Institutions must be built.	
14	Global pollution; small, open economy	Coordination is needed, particularly in competitive sectors.	May be preferable.	Often conflict with trade rules.	Quantity instruments may fit better into framework of international treaties.	International treaties. Money transfers to poor countries.	

Notes: Two-tiered mechanisms include deposit-refund schemes. MBIs = market-based instruments; CAC = command and control; CPR = common property resource; VA = voluntary agreement.

information for provision to consumers, shareholders, and citizens so that they can make educated decisions. In complex cases, the policies needed may be considerably different, depending on the state of the ecology. Before an ecological collapse, the precautionary principle may suggest quantity-type policies (although there is a trade-off with efficiency in case there are many polluters). After an ecological collapse, the costs of recovery and adaptation may be so considerable that subsidies are inevitable in some form.

Row 8 shows the potential for resistance to certain environmental policies by affected agents. Companies, which stand to lose large sums of money in environmental charges, and individual consumers alike may challenge the legitimacy of the policy at hand. Political attention is often focused on policies that might have a regressive effect, so that poor consumers might have larger budget shares for a particular commodity (e.g., some forms of energy). In such cases, careful design of the instrument appears to be crucial for its political acceptability. Tradable emissions permits are sometimes said to be preferable to taxes because the main polluters are compensated by being given property rights, and charges have turned out to be more palatable than taxes in many instances. If polluters share genuine concern for a particular environmental issue, then they are eager to see collected charges earmarked for that environmental cause. For emissions of nitrogen oxides (NO_x) from large combustion sources in Sweden, refunding charges to the polluters makes the net cost of the policy low, facilitates its acceptance, and makes it easier to apply the instrument to only one subgroup of polluters. Established polluters often feel most comfortable with conventional licensing, but licensing may involve significant new source bias. Numerous other aspects concerning the psychology and sociology of policy instruments, omitted from the table because of space constraints, are covered in Chapter 13.

Row 9 reflects the many situations in which monitoring is almost impossible—for instance, the small “emissions” of metals and other toxic chemicals from households. These emissions take place on such a small scale and are so dispersed and intermittent that they are impossible to monitor. They make a mockery of any tax, charge, or physical regulation. The only mechanisms that can work under these circumstances are information (e.g., persuasion or education) and incentives that are self-revealing and self-enforcing (e.g., deposit–refund schemes and certain subsidies). Subsidies and public provision are the best instruments when the environmental issue has a large public good component (as with research into abatement technology or common property resource management). The great advantage of a deposit–refund scheme is that the equivalent of an emissions tax is paid when a product is bought (which is, *a priori*, the only moment when it may be possible to get the potential polluter’s attention). This fee is refunded if there are no emissions, but the burden of proof is placed squarely on the individual, who must return the product to prove that there were no emissions or that there was no inappropriate disposal. This simple example of a revelation mechanism

is central to the design of policy instruments for conditions of asymmetric information (see Chapter 10).

Rows 10 and 11 concern the number of agents and the structure of the markets in question. If there is a monopoly, then policymakers are not going to draft general tax laws to influence that one agent's behavior. It is much simpler to issue a regulation. Furthermore, taxing the emissions of a monopolist is complicated, because there are two market failures: missing markets for pollution and restricted supply by the monopolist. Also, when the agents are few, they may think and act strategically; their chief concern in so doing might not be the market for pollution per se but other markets (e.g., credit, labor, or product markets). Thus, the monopolist company might use regulators to strengthen its competitive position. This risk is obvious for direct regulation, in which more generous regulations for older firms create an indirect barrier to entry for new ones. Similarly, tradable emissions permits can be bought or withheld from sale for strategic reasons. Subsidies are the worst policy option, whereas taxes and charges presumably cause fewer such problems. When the number of victims is small, too, negotiations may be an appropriate instrument.

Row 12 refers to general equilibrium effects and the double dividend issue, which are particularly important for pollutants that are expected to generate substantial long-run revenues (when the optimal pollution level is far from zero). Row 13 is labeled "developing economy," which is too broad a category to convey real meaning. Several considerations generally apply to these countries: low overall income levels, heavy emphasis on distributional issues, and high growth rates (unfortunately, not extant in all of these countries). Many other special features may or may not be common: high population density, ecological complexity, high inflation rates, low effective rates of competition, issues related to corruption, lack of technical expertise, and lack of political support for environmental programs. These kinds of individual country characteristics are more decisive for policy design than the mere fact that a country is one of the many developing economies.

Similarly, Row 14 covers a wide range of issues related to global pollution and global economy issues (see Chapters 13 and 14, respectively). A small, open economy cannot tax productive inputs in the same way that a closed economy can, and this situation may lead to a "race to the bottom" or development of pollution havens unless environmental policies are coordinated in some way. Such coordination is particularly important for global pollutants, for which domestic action without the backing of international treaties is largely symbolic.

Many issues are inevitably missing from the table. With ambient pollution, some special sets of policies exist that encourage collaboration among the polluters because they can help monitor each other. In developing countries, distributional and political economy issues typically are most important, and environmental protection agencies and authorities need to be formed, informed, trained, and financed. Informational instruments and low fees (to

provide funds) might add to the proper mix of instruments in the early stages of policymaking.

Interaction between Policies

It is the nature of policymaking that new regulations and instruments tend to supplement rather than replace older ones. What are the joint effects of using several instruments? Sometimes, several instruments may be needed to address several goals. This is particularly the case in a second-best world. In such a setting, several constraints prevent the attainment of Pareto optimality. If the constraints are market failures, the elimination of only one of them does not necessarily improve welfare (Ben-near and Stavins 2007). Indeed, market failures can be jointly ameliorating (correction of one market failure reduces welfare losses from the other one), jointly reinforcing (correction of one market failure exacerbates welfare losses from the other) or neutral (correction of one market failure does not affect the welfare losses from the other). For example, a substantial body of literature examines the interactions of environmental externalities and market power. These two market failures are jointly reinforced, and the trade-off between social welfare losses that result from decreased production in markets with market power and social welfare gains that result from those same decreases in production in markets with negative externalities are well known (see Chapter 11).

Environmental externalities may also interact with other types of externalities in ways that are jointly ameliorating or jointly reinforcing. A good example is the interaction between positive externalities (spillovers) from technological innovation and negative externalities from pollution. If the outcome of R&D investment is advancement in general knowledge, it can be difficult to exclude others from utilizing the results of R&D investments. This spillover results in under-provision of such investments in the market. While the use of a single environmental policy may create some incentives for technological change, evidence suggests that environmental policy alone does not create sufficient incentives to overcome completely the market failure (see Chapter 9). Combining environmental policies with policies intended to encourage both innovation and diffusion of new technologies—as for example, tax incentives and subsidies—can be welfare enhancing.

The combination of imperfect information and environmental externalities provides another example of a situation where an environmental problem coexists with a market failure. For instance, if we assume that consumers have preferences regarding the environmental impacts of their consumption goods, but the impacts of different consumption goods are not known perfectly by them, we could reduce the environmental problem by providing consumers with such information. To see how they might be jointly ameliorated, consider the labeling and disclosure programs discussed in Chapter 8. These programs

increase consumers' information about the environmental performance of firms and their products' external costs. This information, used by consumers in their consumption decisions, may in turn result in lower environmental impacts.

On other occasions, the use of multiple instruments may reflect historical development or the unintended interaction of legislation at various levels of government (local, regional, national, and international). Indeed, it is rare for a policy innovation wholly to displace existing instruments. Instead the new policy will operate in parallel with existing instruments, interacting with them in a variety of ways. In some cases, there may be legitimate grounds for maintaining other instruments to deliver social objectives other than efficiency, such as political feasibility. Figure 15-1 illustrates such a case in which the optimal level of pollution (E^*) could be achieved (in principle) either by negotiation or through some state-initiated policy, such as taxation (T). However, if the state imposed a tax and the victims then succeeded in negotiating with the polluter, they might arrive at a suboptimally low level of pollution (E^{**}). This combination of bargaining and tax instruments is perhaps not very common. However, many other combinations are likely. For instance in climate policy, subsidies to green technologies may be used to reinforce the effect of carbon taxation (particularly if it was not politically feasible to set the tax high enough). While such a subsidy and a tax may reinforce each other in approximately the intended manner, the same policymakers would find that a green subsidy does not interact additively with a carbon policy such as tradable permits: Once the cap on carbon is set the green subsidy will mainly lead to transfer flows—it will often not reduce carbon emissions any more. The “cap” will also act as a “floor.”

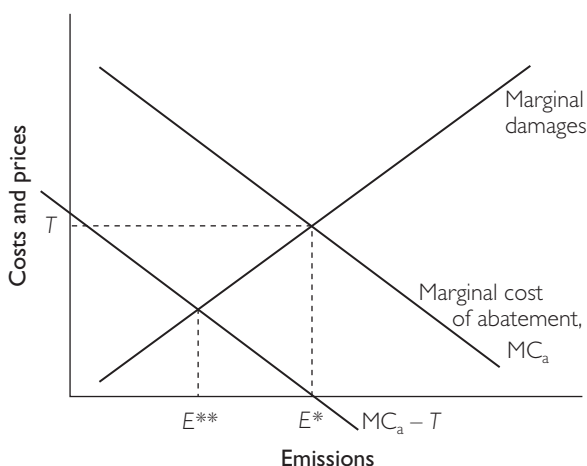


FIGURE 15-1 Interaction between Environmental Taxes and Negotiations

Notes: T = environmental tax; E^{**} = suboptimal level reached through the joint use of two separate instruments (bargaining and tax), each of which would have given the optimal level if used in isolation; E^* = optimal emissions (and abatement) level; MC_a = marginal abatement costs.

Labeling schemes do not necessarily stop or avoid certain industries just because they are subjected to environmental taxation. Similarly, insurance and liability, in various forms, coexist with all other instruments because the payment of environmental tax does not necessarily release firms from responsibility for potential future damages. Permit systems are typically layered over local environmental regulation or other policy instruments. Information disclosure and labeling (possibly separate schemes in different countries) are used and may have market effects irrespective of whether firms pay environmental taxes.

The introduction of climate policies into a crowded “policy space” is likely to bring these issues of policy interaction to the fore (Sorrell and Sijm 2003). Indeed, the EU emissions trading system (ETS) scheme coexists with carbon/energy taxation and support mechanisms to promote renewable electricity and energy efficiency (see Chapter 6). Although the interaction of policy instruments can cause effects that are undesirable, the gradual evolution of policies in a changing environment necessarily requires changes in the policy instruments used, and several instruments inevitably (and probably desirably) will be used during transition periods. Combined instruments require careful analysis of their interactions (Böhringer and Rosendahl 2010; Fisher and Preonas 2010; Goers et al. 2010). Regulators should look for “complementary” policies that preserve the benefits of existing policies to the greatest possible extent and should be administratively feasible at a reasonable cost. For instance, Coria (2011) has analyzed the effects of mixed policies in which tradable permits are used to control the long-run equilibrium level of pollution, and another instrument is used to maintain the pollution below some predetermined threshold during occasional environmental crises that prompt the imposition of more emergency measures to improve environmental quality to a satisfactory level. She shows that the fact that changes in the firms’ constraints induce firms to modify their abatement behavior—and by that means affect the permit price—implies that it is difficult to preserve the dynamic properties of tradable permits by the use of complementary policies.

Practical policymaking is an art of timing, combining, and sequencing instruments to meet multiple goals amidst changing circumstances.

PART IV

Policy Instruments for Road Transportation

THE TRANSPORTATION SECTOR IS IMPORTANT FROM an environmental viewpoint: it is one of the main users of energy and a particularly prominent source of air pollutants: more than 600 million people in urban areas worldwide are exposed to dangerous levels of traffic-generated air pollutants (Han and Naeher 2006).

It also accounts for several other important external costs, such as congestion, accidents, noise, and “barrier effects” (e.g., large roads create barriers to communication and movement of both humans and animals when they cut through a community, effectively making it difficult for people to shop, socialize, or work on the other side of the road). The sector has long been one of the main topics of environmental attention in industrialized countries, and in the past decade, issues of air pollution have taken priority in developing countries, too. In megacities such as Mexico City (Mexico), São Paulo (Brazil), and Santiago (Chile) in Latin America; Ibadan and Lagos (Nigeria) in Africa; and most of the large cities in Asia—Bangkok (Thailand), Bombay and Delhi (India), Jakarta (Indonesia), Manila (Philippines), Seoul (Korea), and many Chinese cities—levels of many air pollutants greatly exceed World Health Organization guidelines and generally are much worse than in the industrialized cities of richer countries. Similar situations exist in many cities of eastern Europe, the former Soviet Union, and the Middle East. Motor vehicles are a major source of this pollution.

Typical values for the vehicle share in total air pollution range from 40% to 99% for carbon monoxide, hydrocarbons, and nitrogen oxides and are somewhat lower for fine particulate matter. The transportation sector is also an important contributor to global warming, acidification, and local air pollution in megacities. In some rural areas of poor countries, runoff and soil erosion due to roads may be significant; water flowing along a hilly road in Kenya or Ethiopia, for example, can erode the soft surrounding soil and may even cause landslides. In heavily urbanized high-income areas, valuable time is lost as a result of traffic congestion; furthermore, congestion increases other costs, such as those related to emissions.

The chapters in Part IV focus on the design of appropriate policy instruments for regulating road transportation, which entails regulating industry. Many of the transport policies actually entail regulating two of the world's largest and most powerful industries: the vehicle and petroleum industries.

By way of exposition, we begin by describing the environmental damage function for transportation. Air pollution from cars consists of many chemical substances that affect human health as well as the environment in general. Each environmental problem results from the confluence of several factors: population density, congestion, weather, type of fuel, vehicle, and driving habits. Sometimes, one main cause of a certain environmental problem is easily identifiable—for example, certain vehicles, a particular fuel, or some other factor, such as unpaved roads or climatic inversion. If so, responsibility for the damage might be assigned to drivers, producers of certain vehicles or fuels, or those responsible for road maintenance. In other cases, it is not easy to separate the effects, hence the responsibility is shared.

Vehicle pollution may be global, regional, or local. Noise is a local problem, whereas smog is local or possibly regional. Carbon dioxide (CO₂) emissions are global, because the negative effects of CO₂ on the environment are cumulative and affect the whole world independently of time or location. Furthermore, because the amount of CO₂ emitted is directly proportional to fossil fuel use, a relatively straightforward policy instrument such as a differentiated fuel tax could be economically efficient; the main problem in implementing such a policy is international coordination. In congested cities, the combination of congestion and local health, attributable to vehicle emissions, makes the design of policy instruments much more complex. The “first-best” instrument might be approximated by environmentally and geographically differentiated road pricing, but practical considerations of transaction and administrative costs might favor some combination of technology restrictions, zoning, fuel taxes, and fuel standards, for instance.

In Chapter 16, we describe the complex damage function for environmental damage caused by the transportation sector and address environmental road pricing as one specific approach to paying for this damage. When it comes to regulating fuel efficiency (see Chapter 17), most European countries have sizable fuel taxes, whereas the United States relies on regulation.

A good deal of policy in transportation is regulation of fuel and vehicles: instruments for fuel quality and the development of biofuels are discussed in Chapter 18. A particularly difficult challenge is to find suitable policy instruments in the megacities of low-income countries, where they are needed the most. Some lessons learned are presented in Chapter 19.

Special thanks to Per Kågeson and Winston Harrington for comments.

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Environmental Damage Caused by Transportation and Road Pricing

In a “first-best” world, it would be optimal to correct for an externality by imposing a correcting charge equal to the short-run marginal external cost or a Pigovian tax. The transportation sector shows how this simple idea can be complicated because the damage function is so complex. The global, regional, and local components of the damage require different logic. In this chapter, we describe the damage function and discuss how it can be sufficiently simplified and stylized to serve as a starting point for the discussion of policy instruments, such as road pricing. Political interest in using pricing within the transportation sector appears to be increasing, as reflected in a green paper on fair and efficient pricing in transportation (European Commission 1995). Whereas no country has yet been able to implement advanced environmentally differentiated road pricing, some sophisticated examples of road pricing, area licensing, and mileage taxes include advanced traffic-management schemes in Singapore, toll roads in Norway, congestion fees in London and Stockholm, and a road pricing scheme in Switzerland that uses the Global Positioning System (GPS). The main reason for this interest in road pricing—besides a general increase in environmental awareness—is presumably the fact that modern information technology has made various road-pricing systems more realistic options.

Vehicles

The main global damage associated with transports is the emission of carbon dioxide into the atmosphere and the consequent climate change. As discussed in Chapter 14, climate policies have many complexities, notably, coordination among countries, the inclusion of all the relevant climate gases and polluting sectors (see Box 16-1 on climate change policy and aviation), and the role of sinks. However, in the context of determining the damage caused by driving a mile in a vehicle, global damage is the least problematic because it depends on only the fossil carbon content of the total fuel consumed.

BOX 16-1 Aviation and Climate Change

According to IPCC (2007) estimates, aviation currently accounts for about 2% of human-generated global CO₂ emissions and approximately 3% of the potential warming effect of global emissions on climate change, 80% of which is attributed to commercial aviation. Compared to the entire transportation sector, which generates approximately 20% of all anthropogenic GHG emissions, aviation's share of emissions is small but steadily increasing. Indeed, despite its relatively short history, commercial aviation has undergone tremendous growth. Since 1960, passenger traffic has grown nearly 9% per year, although this rate has decreased slightly to 5% since 1999 as the industry began to mature.

Aviation affects climate change not only through the emissions of CO₂ but also through the creation of high-altitude ice clouds formed in the wake of the aircraft. High clouds increase the amount of solar radiation that is reflected. As a result, they have the potential to influence climate on a global and regional scale. The latter effect has been found to potentially play a major role in climate change accounting for the largest impact from the aviation sector (IPCC 1999). The negative environmental effects, in addition to the forecast growth of this sector, makes aviation the most unsustainable mode of transport currently available.

Even in the context of future technological and operational improvements that will continue to decrease the emissions from aviation activities, experts believe that these improvements alone will not be sufficient to counterbalance the increase in emissions generated by the growth in global air traffic. Further action is therefore required in the form of policy interventions. However, the policy design must take into consideration the mobile nature of the emission sources, the international character of the activity, and the fact that the total impact on climate change is wider than the direct effects of CO₂ emissions—and thus an overemphasis on CO₂ reduction might lead in the long run to the development of technologies with significantly higher emissions of other pollutants to compensate for the reduced CO₂ emissions.

So far, no global agreement on the reduction of aviation emissions has been reached. Nevertheless, the European Union included aviation within the EU emissions trading scheme (EU ETS) in October 2008 (Directive 2008/101/European Commission). From 2012 onward the EU ETS will require compensation for emissions on domestic flights within the European Economic Area (EEA), flights between EEA states, and flights between EEA States and countries outside the EEA (third countries). The program's objective is to reduce the impact on climate change of aviation emissions and to ensure that the aviation sector contributes to the achievement of the overall objective of the Community: a 20%–30% reduction in emissions by 2020 compared to the 1990 level. The project is still in its initial stages: Throughout 2010, operators must monitor and report ton-km data, which is required for the allocation of free allowances. The European Commission was scheduled to publish total number of allowances and benchmarks in September 2011, with the final allocation and transfer of the emission allowances to operators' accounts expected to occur in February 2012.

The inclusion of aviation in the EU ETS system is expected to generate changes in air traffic volumes (albeit very small), incentivize technological changes for air transportation (engine and combustor designs, lighter aircraft construction materials, more aerodynamic aircraft designs), and encourage other transportation modes due to substitution effects. The economic effects of the inclusion will mostly be felt by the final consumers of aviation services, since all additional costs will most likely be

passed forward from the airline companies to the consumers, as happens when all competitors in a market are covered by the EU ETS (e.g., the EU energy market). In this case, a decrease in demand is expected, both for cargo demand and passenger demand, albeit estimated to be small, especially due to the ability of the aviation industry to price discriminate between consumers. With regards to this last point, many short haul flights could be replaced by inter-city rail where the impacts per passenger kilometer can be much lower. Nevertheless, this might require railway infrastructure investments and better integration with airports. In the case of freight, shipping could be an alternative. Shipping is often recognized as a sustainable, energy efficient and “relatively” environmental friendly mode of transport, although it is still a source of GHG emissions. Other environmental problems are associated with shipping—vessel oil spills, ballast water disposal, waste disposal at sea, and so on.

Regional problems such as acidification differ considerably with respect to the location of emissions because the current pollution pressure and the sensitivity of the ecosystems, soil, and underlying rock vary dramatically (see Chapter 20); within a region, the differences are typically smaller. However, the fuel and engine characteristics that give rise to various emissions differ considerably.

Local externalities include congestion (see Chapters 2 and 12), noise, and air pollution. The modeling of the damage from pollutants must account for several complicated technical, atmospheric, chemical, ecological, and health aspects. First, the vehicle emissions contain thousands of chemicals and may vary considerably depending on fuel and vehicle characteristics (described below). The pattern of atmospheric reactions and the transport of these pollutants involve many factors that depend crucially on the weather as well as the topography of the city. Therefore, translating emissions into ambient levels of pollution is complicated. The next level of complication is estimating the damages (to health, ecosystems, and capital) caused by these levels of pollution. The damage depends largely on population density. The dose–response functions for human health are complicated by the long-run gestation of many of the induced medical conditions. Effects also vary with respect to timing (e.g., problems related to tropospheric ozone are highest during the summer, and noise is worst at night). Ultimately, these damages then have to be valued, which introduces yet another series of difficulties. Translating mortality, morbidity, suffering, and days of work lost due to sickness into monetary values presents methodological as well as ethical problems. Various methods—stated preference, revealed preference, and simple loss of production values—have been used in the literature.

For the purposes of this discussion, it suffices to state that these damages vary dramatically with respect to the exact time and location of the emissions. The health costs depend on the number of people affected and are considerably magnified in a city center; location and timing can account for differences of several orders of magnitude. In addition, emissions exhibit a similar range

of variation among vehicles. When these factors are multiplied, it becomes apparent that the cost of driving a dirty vehicle (with poor exhaust characteristics) one mile in a city during rush hour is different from that of driving a relatively clean vehicle one mile in the country. Environmental damages also depend on such factors as fuel choice, driving style, and weather conditions. When there is heterogeneity in damage and abatement costs, the risk of incurring large costs by inappropriate design of instruments is the highest (Chapter 9), and in the case of transportation, both types of heterogeneity exist. Damages incurred by driving one mile vary strongly by location, suggesting the need for geographical (and temporal) differentiation.

Abatement costs also vary because they incorporate the costs of driving less often, scrapping cars, retrofitting vehicles, and similar actions—the costs of which vary enormously between vehicles. As a result, the potential gains from market-based instruments are large¹.

Location

Damages to health often make up a large percentage of local pollution costs. Exhaust emissions tend to disperse quickly, which makes the exact location of emissions a crucial factor in determining the extent of damage. Driving in an area where the density of exposed population is high results in higher environmental damage than driving in an area where few people are exposed to emissions². This factor can easily vary by several orders of magnitude—hundreds or thousands of times.

Other geographical and climatologic conditions are equally important. Many of the worst affected urban areas have distinctive features, such as high altitude or being surrounded by mountains (Mexico City, for example). Table 16-1 presents an estimate of the importance of geographical location for Gothenburg, Sweden (population roughly 0.5 million). The values build on conditions that include population density, climate, fuel, vehicle stock characteristics, and willingness to pay for averting respiratory diseases—which are related to income levels and many other specific conditions. The exact data are not transferable to other contexts, but the values do reflect large differences

TABLE 16-1 Environmental Values and Geographical Location

<i>Pollutant</i>	<i>Regional environmental effects (US\$/kg)</i>	<i>Local environmental (health) effects (US\$/kg)</i>		
		<i>Country</i>	<i>City average</i>	<i>City center</i>
VOCs	1.7	0	5	25
NO _x	4.0	0	5	25
PM	0.0	18	90	450

Notes: VOCs = volatile organic compounds; NO_x = nitrogen oxides; PM = particulate matter. Values were converted at 10 Swedish krona = US\$1. *Source:* Johansson and Sterner 1997.

in health values in the inner city, the suburbs, and the country. This variation is derived mainly from the differences in the number of exposed people and might be similar in similar towns. In large, densely populated cities, the differences may be greater³.

Combining Vehicle Age and Location

The power of technical progress in the area of engine and exhaust technology is considerable. Table 16-2 presents one estimate of the effects of technical progress on vehicle emissions by model year. Because road-pricing systems and other policy instruments are designed to work for a relatively long time, it is important to consider expected future emissions. Again, the important point is not so much the values per se but their approximate rates of change.

The values on the left side of Table 16-2 show which emissions rates may reasonably be expected of new vehicles in each respective class and model year. From the 1988 cars to the 2000 cars, emissions of volatile organic compounds (VOCs), nitrogen oxides (NO_x), and particulate matter (PM) improved on the order of 5–10 times. In previous decades, similar progress (broadly speaking) took place; the advent of the catalytic converter alone resulted in reductions of one to two orders of magnitude for many pollutants. Similarly, future emissions will be much lower than the present ones⁴. Within vehicle model years, another difference is normally found between individual vehicles and the actual vehicle stock. At any one moment, many vehicle model years are on the

TABLE 16-2 Estimated Emissions for Different Vehicles

<i>Vehicle model year</i>	<i>Estimated average emissions</i>		
	<i>VOCs (g/km)</i>	<i>NO_x (g/km)</i>	<i>PM (mg/km)</i>
<i>Passenger cars, gasoline</i>			
1988	2.50	1.53	37
2000	0.46	0.17	7
2010 (Euro 5)	0.1	0.06	5
2014 (Euro 6)	0.1	0.06	4.5
<i>Buses, diesel</i>			
1988	1.30	13.2	500
2000	0.40	7.3	150
2010 (Euro 5)	0.295	0.235	5
2014 (Euro 6)	0.195	0.105	4.5

Notes: Values are corrected for cold starts, climate, driving cycle, and deterioration of emissions reduction systems over time. Emission factors for 1988 and 2000 are based on Ahlvik, Egeback, and Westerholm 1996. Emission factors for 2010 and 2014 are based on Euro 5 and Euro 6 standards, which come into force in the respective years, and to which newly manufactured vehicles must comply.

road, so the overall difference in emissions characteristics (and thus environmental damage) is much greater than that shown in Table 16-2—probably three orders of magnitude (1,000:1) when comparing the dirtiest vehicles to cleanest vehicles in industrialized countries. The divergence between new and old cars can be expected to be even bigger in poor countries because of the tendency to keep cars that would be scrapped in richer countries (and even to import old cars from other countries).

Engine Temperature and Other Factors

Many engine parameters are important for the rate of emissions. One is engine temperature, which has a large effect in cold climates, because combustion is much less complete before the engine and the catalytic converter have warmed up (Table 16-3). In cold climates, a significant share of total VOCs and carbon monoxide (CO) is typically emitted during the first kilometer driven, even of a fairly long trip⁵. Because most urban trips are fairly short (often averaging under 5 kilometers), cold start–related emissions typically make up more than half of total emissions for VOCs and CO (but not for NO_x). These emissions can be reduced by using cleaner fuels, preheating the engine or the catalytic converter, or using faster catalytic converters. Considering the importance of this factor, special attention may be warranted because it is unlikely to be picked up by other policy instruments that policymakers choose.

Similarly, several other factors (e.g., weather, fuel, driving style, and congestion) have a strong influence on total environmental damage. The most notable weather factor is wind (maybe a factor of at least 3, according to Leksell and Löfgren 1995). In some locations, patterns of weather known as thermal inversion cause pollutants to accumulate over several days, reaching very high levels—cool air traps a warmer bubble of air over a city so that the air cannot disperse. Fuel quality (e.g., chemical composition, the use of reformulated fuels, and the addition of alcohols and ethers) can have substantial health effects, which are strongly reinforced during periods of thermal inversion. One striking example of the importance of fuel composition is the use of lead additives in gasoline (see Chapter 18). For driving behavior, significant environmental improvements are possible (Rouwendal 1996), and good logistical planning in transportation companies could cut fuel use (and emissions) considerably.

TABLE 16-3 Effect of Temperature on Emissions for a Car with Catalytic Converter

Outdoor temperature (°C)	First kilometer driven		Second kilometer driven		Warm engine	
	VOCs (g/km)	CO (g/km)	VOCs (g/km)	CO (g/km)	VOCs (g/km)	CO (g/km)
22	2.6	21.0	0.07	0.16	0.02	0.12
7	15.7	123.1	1.38	11.0	0.25	0.80

Source: Laurikko, Erlandsson, and Abrahamsson 1995.

Congestion causes a double effect. First, the time cost of a vehicle mile rises rapidly with increased congestion, because the addition of a vehicle to an already congested network increases travel time for many other passengers (see Chapter 2). Furthermore, this effect tends to interact with emissions: because the average speed is reduced to levels that are far below the optimal operating speed for vehicles, the rate of emissions per mile increases as well (Johansson 1997a)⁶.

Calculating Environmental Damage from Road Transportation

The cost that is relevant for a Pigovian tax is the sum of damages incurred (see Chapter 4). For a certain vehicle, this cost is not easy to calculate because—as discussed previously—many complex processes govern emissions and because of the causal chain from emissions to ambient pollution, exposure, and damages. To simplify the calculation, a stylized damage function is used:

$$D = D(e, g, t, w) \quad (16-1)$$

where D is damage, e is vehicle emissions, g is location, t is time of day, and w is weather. Vehicle emissions are equal to distance (m) times emissions rate (ξ), which is a function of vehicle characteristics (v), fuel characteristics (f), outside temperature (to), road conditions (o), and a vector of driving-related variables (z) that include speed, vehicle maintenance, acceleration patterns, and engine temperature [$\xi = \xi(v, f, to, o, z)$]. In practice, one of the main determinants is congestion, which determines average speed. A perfectly differentiated Pigovian tax like this is not possible, but some form of environmentally differentiated road pricing might be (see Box 16-2).

For an instrument such as the one in Box 16-2 to work, the driver must have knowledge of the current fee, and so the charges are assumed to be calculated and displayed on a device rather like a taxi meter. All the components for this kind of system are currently available at a cost that would not make a significant difference to the price of new cars. Many of these kinds of equipment would be cheap if produced in large quantities.

In many industrialized countries, the auto industry has already introduced GPS or similar systems for other reasons—as a service that can offer drivers information about best routes and parking. GPS monitors are fairly inexpensive and becoming standard on private boats and in mobile phones. When we first wrote about this technology it was science fiction, but today many taxis and truckers have years of experience with this kind of system. GPS satellite navigation enables drivers to easily reach any destination, even in a new city or country. This system also saves fuel, time, and money by emphasizing good logistics: minimizing search and route selection time. With this equipment, the car already has the hardware necessary for the type of charge system described in Box 16-2. All that is needed is a program to calculate the fees.

BOX 16-2 A Feasible Scheme for Environmentally Differentiated Road Pricing

As an illustration, consider a simple multiplicative fee structure that approximates Function 16-1 as a variable tax (T) per mile:

$$T = v \times g \times t \times w \quad (16-2)$$

In an intermediate system, vehicle characteristics (v) could be a number from 1 to 10, where 1 is the least-polluting vehicle (which might be electrical, for instance, but still causes wear and tear of roads, congestion, and thus some other externalities—indirectly at least) and 10 is the most-polluting vehicle. (Actual emissions rate could be measured with on-board diagnostics, but for simplicity, it is replaced here by v , which would be based on make, age, and periodic inspections or standardized tests.) The rating “ v ” might possibly be contingent on the use of a special fuel, because, for instance, diesel engines have different emissions characteristics when running on alcohol or on different kinds of conventional or reformulated diesel fuel.

Geographical location (g) might range from 1 (in the country) to, say, 10 or 100 in a city center. This variable might seem to be difficult to monitor for mobile sources but actually can be quite cheaply done by using Global Positioning System (GPS) or even telephony or transponders located along the roads or at toll booths.

Time (t) might have only two values, such as 10 for rush hour and 1 for any other time, or it might be more directly related to a measure of current congestion. Similarly, for weather (w), a high value might be reserved for special conditions such as thermal inversion or smog episodes. This variable could be broadcast over the radio—making it possible for authorities to limit traffic (by raising the fees) on exceptionally bad smog days without actually banning traffic.

Potential problems include issues of privacy, tampering, and the period of “transition” before all vehicles have this equipment. In terms of privacy, some would find monitoring the locations of all vehicles unacceptable (suggesting associations with a “Big Brother” society), but doing so is not necessary for such a system to function; it is sufficient for each vehicle to know its own position (the information does not need to be stored centrally). The issue of tampering with such devices must be addressed because the whole scheme hinges on the correct functioning of a GPS and software that incurs considerable cost to motorists. Monitoring could be difficult in countries or regions where inspection programs are inadequate. The severity of these issues will depend on the construction of the tariff system. The mechanism must be designed so that the compliant motorists who have clean vehicles and maintain them properly are rewarded.

The calculation in Equation 16-2 (in Box 16-2) is a simplification. It does not account for the environmental characteristics of different types of fuel, the disturbance caused by noise from traffic at night in residential areas, the environmental effects of speed (on emissions and external risks), and the particular risks of driving in sensitive areas. However, the potential advantage

of incorporating such variables into the equation might not warrant the additional complexity of the instrument that would be required to do so.

Some of these variables are addressed separately as, for example, speed limits and no-traffic zones and as quality requirements and specifications for fuel. If possible, it is preferable to linearize the damage function and use one instrument for each subgoal. Unfortunately, it is not always possible when the damage function is truly nonlinear, as in Equation 16-2. If one instrument is used for the location factor and another for the vehicle, then the strong interaction between the two variables is lost. The effect will be restrictions that are much too tough for vehicles driven in the country and for urban vehicles that drive short distances, but much too lax for vehicles that are driven heavily in city centers (e.g., taxis, buses, and delivery vans). Practical policies will have to trade off the costs of complexity and the suboptimality of simpler fee systems.

Simpler Pricing Schemes

It is not possible to build a single monolithic tariff system that relies on all vehicles having all the relevant equipment before the system can operate. No town in the world has yet implemented full-fledged environmental road pricing, and in fact, it may be too expensive to obtain all the potential cost savings because of the high costs of information, administration, and transactions⁷. Therefore, the instruments that are available in the transportation area—some of which are simplifications of a fully developed environmentally differentiated road pricing scheme—should be considered a useful benchmark for comparison. In this chapter, we present various schemes for road pricing, including area licenses, mileage taxes, and road tolls; we also discuss differentiated vehicle taxes⁸.

Considerable health and environmental benefits can sometimes be gained with simple regulatory measures such as phasing out leaded gasoline, retiring the most-polluting vehicles, paving roads, rehabilitating public transport, managing traffic, or enhancing driver education. (For a discussion of such priorities in developing-world megacities, see Chapter 19.) One example of the system costs of sophisticated road pricing that probably exceeds potential savings is for cold starts. Obviously, a system would have to be complex to distinguish the first kilometer driven, when the catalytic converter is cold and emissions are high. This monitoring could be done with on-board instruments, but it would be costly. Technological fixes (e.g., the more widespread use of electric preheating of the motor and/or catalytic converter) probably can solve this problem more efficiently. Presumably, administrative ways of encouraging and mandating such solutions might be simpler and cheaper than building the variables into a road-pricing model⁹.

Another example of system costs exceeding the potential savings is global warming, where environmental damage does not depend on the time or location a vehicle is driven but on the original source of the vehicle's fuel.

On-board systems for road pricing would have to be extremely sophisticated to differentiate between fossil methanol and biomethanol, so a simple tax on the fossil carbon content of the fuel is a superior instrument for this situation.

A third example of the cost–benefit trade-off is the protection of schools, hospitals, and residential areas from excessive levels of pollution and night-time noise. Zoning and regulation are probably the most efficient approaches to this kind of problem.

Area Pricing

One simple road-pricing scheme is the area pricing system, $T = f(g)$, or possibly a more sophisticated system that differentiates by time of day and vehicle characteristics, $f(g, t, v)$, which mainly targets the problem of congestion and pollution hot spots. (For a general survey of theory and applications of road pricing, see Parry and Small 2009.)

An advanced system is the satellite-based GPS system, which allows the charges to be related to distance (and varied between zones in the city). Clearly, this system is more efficient since it charges vehicles per mile driven within the zone instead of just per entry or per unit of time. Nevertheless, it is also more costly to implement. At the other end of the scale are simple systems that, for instance, require all vehicles in a certain jurisdiction (such as a town) to display a valid pass on the windshield. Although this blunt instrument does not account for the number of miles traveled, it can easily be differentiated by vehicle characteristics. In principle, such passes could reflect environmental characteristics, even if to date, the systems mainly distinguish broad categories (e.g., cars, buses, and trucks). Finally, intermediate systems might consist of semiautomatic road tolls in a ring around a town. The choice between systems must be based on the trade-off between environmental effects and administrative costs. In such a case, the simplest and the most advanced systems may be superior to the intermediate ones.

In practice, the concept of area pricing is spreading all around the world. Schemes that differ in both design and outcomes have been implemented successfully in a wide range of cities—Singapore (1975, see Box 16-3), Bergen (1986), Oslo (1990) and Trondheim (1991) in Norway, Rome (2001), London (2003), and recently Stockholm (2007). A range of smaller cities have also implemented congestion pricing schemes, such as Durham (England), Znojmo (Czech Republic), Riga (Latvia), and Valletta (Malta). (See GTKP 2009.)

In London, the scheme is intended to reduce congestion in the central area (approximately 40 square kilometers) and raise funds for public transportation. Vehicles are subject to a fixed £8 congestion charge if driven or parked within the charging zone between 07:00 and 18:00 weekdays. Since the system is designed to reduce congestion, the fee must be paid in advance, either online, by mobile text messaging (SMS), or many other methods). Cameras at entrances, exits, and around the zone read the number plate using

BOX 16-3 Transport Management in Singapore

In Singapore, which has a population of more than 4 million people in an area of only 700 square kilometers, traffic and air pollution have been serious concerns since a long time ago. By the early 1970s, the motor vehicle population was near 300,000 and traffic congestion was leading to urban strangulation (Olszewski, 2007). An Integrated Land Transportation Strategy was adopted in the mid 1970s pursuing the following aims:

- Integrating land use and transport planning,
- Increasing road capacity as much as feasible
- Providing an efficient public transport system
- Managing demand for travel, especially that from private cars.

In order to reach the last target, several ownership restraints have been introduced over the last three decades, as for example, a variety of taxes and registration fees, direct quota imposed on the number of new vehicle registration, gasoline taxes, parking fees, an Off-Peak Car Scheme and road pricing.

- Under the Vehicle Quota System, potential car owners must purchase a vehicle entitlement at a monthly auction.
- The Off-Peak Car Scheme (originally called the “Weekend Car Scheme”) was introduced to enable people to own cars while avoiding congestion. Newly purchased cars could be registered as off-peak cars with an upfront saving on reduced registration fees. To facilitate enforcement off-peak vehicles are identified by red license plates.
- When it comes to road pricing, initially motorists entering the Central Business District (CBD) during peak period (7:30–10:15 AM) had to buy a license. However, as the coupon system became too cumbersome, it was upgraded to a fully automated Electronic Road Pricing in 1998 where motorists pay electronically and avoid toll delays. The system charges motorists for road use according to vehicle, time of day, and target level of congestion. A method called “shoulder pricing” is used, which implies increasing the fee in steps every half an hour before the peak and decreasing it after the peak. The current rates for cars are:

7:30 – 8:00 AM \$1.50

8:00 – 8:30 AM \$2.00

8:30 – 9:00 AM \$3.00

9:00 – 9:30 AM \$1.00

Nevertheless, in order to ensure smooth traffic flow a five-minute transition rate is used whenever the rate increment is \$ 1.00 or more. Thus, for example, \$ 0.8 is charged from 7:30 to 7:35 and \$ 2.50 from 8:30 to 8:35 (Olszewski, 2007).

The congestion fee has had a significant effect on reducing traffic. Immediately after its introduction in 1975, car traffic entering the city center during the peak decreased by over 70%. In spite of the rapid economic growth in Singapore observed during the last decades and the increased people’s desire to own and use a car, the car share of work trips stays still below 25% (see Santos 2005 for a comparison between congestion pricing in London and Singapore).

an automatic computer system and verify whether the fee has been paid. If a match is found, any images of the vehicle from the database are automatically deleted. If no match is found, a penalty fee of £120 is issued.

The London congestion tax is the first important experience and the best known success in reducing congestion in European big cities. Since its introduction, the fee has increased road traffic safety, reduced traffic in Central London by almost 30%, and cut down wait times at bus stops and air pollution (Leape 2006). Overall benefits of the fee are estimated to have exceeded its operating costs by a ratio of about 1.7 (Transport for London 2008). The revenues of the program have been used primarily to improve the public transportation network and build roads, bridges, and walking and cycling facilities.

Following London's successful experience, the city of Stockholm implemented in 2007 a time-differentiated congestion tax scheme designed to reduce congestion, increase accessibility, and improve air quality in the inner city. The charges are placed in the entrance of the city center in the form of two cordon lines where vehicles are charged every time they cross. The charge is very low compared to London, varying according to the time of the day between 10 SKr (approximately 1 EUR) and 20 SKr (2 EUR), with a maximum daily charge per vehicle of 60 SKr (6 EUR). No charge applies on weekends. Vehicles are registered and identified automatically at the 18 control points, and payments are due monthly after invoices are sent to registered owners of the vehicles.

Results of this scheme are more than satisfying so far (Eliasson 2009). Traffic over the tax cordon was reduced by 18% after the system was implemented, accidents were reduced an estimated 3.6%, and use of public transit expanded: approximately 50% of journeys switched to public transportation (City of Stockholm Traffic Administration 2009; Kottenhoff and Brundell Freij 2009).

Thus, even though urban congestion pricing schemes were initially considered unfeasible due to the difficulty of establishing the optimal charge and the high operating costs associated, such systems have become feasible solutions to the ever-increasing traffic congestion in large urban agglomerations, mainly due to advancements in optical character recognition technologies. It is interesting to see the power of examples: the congestion fee was long overdue when introduced into a megacity like London, which suffered serious congestion for decades. Then in just a matter of a few years the scheme started spreading to much smaller towns because the technology has been proven and maybe also because the instrument has withstood the most difficult test: that of political acceptability. In most cities it has been very controversial and fiercely opposed at first, but then quite quickly become popular or at least acceptable.

Mileage Taxes and Road Tolls

Two common road-pricing instruments are mileage taxes and road tolls. They typically have fees per mile [$T = f(m, v)$] that may or may not be differentiated by various characteristics of the vehicle or other factors. Their simplicity

keeps down the cost of monitoring and fee collection (two of the main arguments why roads should be public goods). In several locations worldwide, vehicles that travel certain toll roads are equipped with a transponder whose personalized signal is picked up when the car passes through an intersection. Enforcement is accomplished by a combination of a camera which takes a picture of the car and a radio frequency keyed computer, which searches for the driver's transponder and then relays data to a central computer; it then calculates the charge and sends a bill automatically to the vehicle's owner. The system also sends a notice and fine to cars that pass through without having an active transponder or paying the toll.

Mileage taxes are a national equivalent of the road toll, but they do not have any particular point of collection. They apply to all roads and are assessed on the basis of periodic readings of vehicle odometers.

The environmental effect varies depending on the type of road tolls. Tolls (on existing congested roads) can be used to reduce congestion and pollution in an area, but this role must be distinguished from the financing of new roads¹⁰. In the United States, congestion is a major cost in itself. Drivers in California's most congested urban areas have suffered an annual loss of more than US\$14 billion in excess time and gasoline. Congestion costs the average driver in San Francisco, California, around US\$1,000 in addition to substantial environmental damages.

Mileage taxes and road tolls can, at least in principle, be differentiated with respect to environmental aspects. Usually, road tolls are based on the size or weight of the vehicle, which is an important factor determining the wear and tear on roads as well as the level of emissions. Since 2001, German expressway user charges are differentiated not only by time but also by exhaust characteristics, making it cheaper for trucks that meet higher environmental standards to operate. This program is called Emissionsbezogen Autobahnbenutzungsgebühr (Bundesamt für Güterverkehr 2002).

Interestingly, the GPS is being used to register truck mileage in a new fee-collection system in Switzerland [$T = mf(v)$]. This fee is part of an agreement with the European Union to increase the permissible weight of trucks. Switzerland has a large share of transit traffic and a strong local opinion against them. The fee is applicable to all trucks that weigh more than 12 tons, on all Swiss roads, and is paid per ton, per kilometer¹¹. They are differentiated by three emissions categories (see Table 16-4) but not by geographical zones within Switzerland.

Differentiated Vehicle Taxes

Environmentally differentiated vehicle taxes [$T = f(e)$ or $f(v)$] are an intermediate option between road pricing and the pure command-and-control strategy based on emissions standards. Taxes are usually levied on vehicle sales (especially new vehicles) as well as annual registration fees.

TABLE 16-4 Fees in the Swiss Heavy Vehicle Fee System

<i>Emissions category</i>	<i>Fee (cents/ton/km)^a</i>
Category 1: Euro 0, I and II	3.65
Category 2: Euro III	3.16
Category 3: Euro IV, V and VI	2.69

^a Measured as cents per ton of vehicle weight per kilometer driven (based on a conversion from Swiss Franc to 2011 U.S. dollars). *Note:* For more information on the categories (European regulations for new heavy-duty diesel engines), see DieselNet 2011.

In Sweden, cars with catalytic converters were given tax credits as early as 1986, and starting in 1992, sales taxes and annual taxes for vehicles were differentiated according to three environmental classes that depend on emissions factors (one of which was the legal minimum). However, the European Union objected to the Swedish differentiation of sales taxes, and the system was abandoned. Currently, there are two standards for passenger cars, and the strongest standard (Environmental Class 1, from 1995) is exempted from annual tax on car ownership for the first five years. Such classes may also be used for regulatory and other purposes (see Chapter 17).

Several arguments favor a system of taxes based on environmental classes for vehicles. First, it creates incentives for dynamic efficiency, that is, for car manufacturers to make cost-efficient improvements beyond current emissions standards. Although theoretically less efficient than perfect road pricing (because it does not specifically target vehicles with high mileage or vehicles that drive in cities), such a system provides incentives at a modest administrative cost.

Second, such a system may be part of a “second-best” strategy because of incomplete and asymmetric information. The authorities have limited information concerning the costs of tougher emissions standards, and environmental classes may be a way of obtaining such information. If the difference in tax between the environmental classes is sufficient to create an output response from manufacturers, then the additional cost is less than or equal to the tax difference. This kind of information is crucial for future policy decisions.

Third, the same environmental classification (extended with classes for older vehicles) can serve also for road-pricing purposes. Fourth, such a system would provide incentives to environmentally aware consumers and to companies with a “green” image to buy environmentally friendly (maybe green-labeled) vehicles.

Finally, a large share of new vehicles are bought as company cars. A few years later, they typically are sold and become private cars. The buyers of the used cars have much less influence on environmental characteristics than the original buyers. Policies that can affect the environmental image of the companies buying company cars thus have an important effect on the whole fleet. Most countries, at least in Europe, have different sales taxes for new cars; cars are frequently differentiated by weight and sometimes by emissions category. However, the differentiation is rarely strong enough to have a significant influence.

Environmental differentiation of the sales tax of new cars might be more effective than a corresponding differentiation of the annual taxes because the latter are implicitly discounted at a higher-than-market rate of interest. Because new cars are thought of as luxury items and bought by people with higher-than-average incomes, some people may find the high excise tax warranted. However, such taxes may be detrimental from a strictly environmental viewpoint. Old cars typically are more polluting; hence, to decrease air pollution, it is desirable to speed up the turnover rate of the vehicle fleet (as is consciously done in Japan).

During the last couple of years, we have witnessed very strong policies in favor of vehicles that are by varying definitions “environmentally friendly.” In Sweden both national and local authorities have been supportive in many ways: subsidies on greener fuels (such as ethanol), free parking, exemption from the congestion fees in Stockholm, lower vehicle purchase taxes, local annual registration taxes, lower private benefit taxation of company cars, and sizable cash subsidies on top of all this. The total value of the subsidies could be up to US\$ 5000/year and, as analyzed by Kågeson (2009), a very expensive way of reducing carbon emissions (at least as long as one does not attach a high value to the dynamic effects of speeding up the introduction of newer technology).

A Gradual Evolution of Road Pricing

One problem with advanced schemes for environmental road pricing is their complexity. Because the damage function is complicated, the optimal tax also would be complicated. Ideally, such systems should be introduced in a large area such as the European Union¹². It would need modern equipment, such as GPS, which only some vehicles would have. However, including the older, more polluting vehicles that normally do not have any such fancy equipment is particularly important. The system also must be flexible enough to detect tampering efforts and to account for temporary visits by vehicles from outside the jurisdiction. Because real policies evolve within a given context and have to be acceptable to many stakeholders, the best way to introduce an advanced system of taxation with this kind of complex control technology may be to build on simpler systems first.

It is difficult to imagine how motorists, town planners, and the vehicle and oil industries could be forced to collaborate on a new system that requires considerable investments in new technology if they totally resisted the idea. Many people are opposed to the instruments that are most efficient. In one study, a few thousand randomly chosen citizens (half of whom had cars) were asked to rank various measures for reducing vehicle pollution. The most popular were a ban on private cars in city centers; technical requirements on motors or fuel, even to the extent of allowing only electric cars in city centers;

road tolls; and speed limits. The least popular were higher taxes for larger cars, carbon taxes, and local taxes on mileage (Bennulf et al. 1998). The fact that the most efficient instruments are the least popular certainly poses a challenge to economists to be more pedagogical. It also reflects the importance of the distributional effects of taxes.

To introduce road pricing gradually, policymakers must allow for the parallel operation of different fee systems. It is simply not possible to introduce a completely new differentiated system of road pricing overnight. One introduction strategy is to make it attractive for some group of motorists to reveal themselves as less polluting by voluntarily joining a more advanced fee scheme. Because less-polluting vehicles are likely to have more sophisticated equipment, the new, more sophisticated fee systems must be made attractive to the drivers of those vehicles. Environmental road pricing would lead to less opposition if it were formulated not as new taxes on vehicles that pollute more, but as exemptions for vehicles that pollute less. This is one way of using “self-revealing” policy instruments (see Chapters 9 and 10).

For example, in areas where motorists already are charged high fees per day or per mile for city traffic, pay high parking fees, or are subject to other vehicle-related restrictions, fee reductions may be attractive to drivers who can show that their vehicles are cleaner and cause less environmental damage than many other vehicles. Following this line of reasoning, policymakers might exempt really clean (e.g., hybrid or electric) vehicles from such fees in cities that have schemes that otherwise charge per-day fees. Similarly, vehicles that are very clean (but not electric) might be (partly) exempted from mileage taxes, road tolls, parking fees, or other fees, and vehicles that use nonfossil fuels might have fuel taxes reduced to reflect the vehicle’s environmental benefit.

Vehicles with certified odometers that can prove they drive only short distances could be eligible for reduced vehicle taxes or refunds on fuel taxes (to compensate for low-income people in rural areas, where the externalities of driving are small). Vehicles with GPS and certified odometers that could prove they drive only short distances in urban areas would have the possibility of paying for the miles they drive instead of paying an area-wide fee¹³.

Supplemental Reading

Button and Verhoef 1997	Proost and Van Dender 2008
ETTM 2002	Santos 2004
Goodwin et al. 2004	Santos and Fraser 2006
Hakim, Seidenstadt, and Bowman 1996	Verhoef and Small 2004
Lewis 1993	Vickrey 1963
Parry 2009	Viton 1995
Parry and Small 2009	Walters 1961

Notes

- 1 Consider, for example, the cost of requiring all car owners in India to retrofit their cars with catalytic converters to curtail urban smog in Calcutta.
- 2 In fact, even the population characteristics may be important. Children, pregnant women, and people with certain ailments or illnesses are typically more sensitive to pollutants; thus, the damage would be higher near a maternity ward, hospital, or school than when driving elsewhere.
- 3 Many cities in the United States, Canada, and other countries have low population densities (i.e., 10 people/hectare compared with around 50–70 people/hectare in many European cities and more than 100 people/hectare in Tokyo, Japan). In U.S. cities, the environmental damage per mile driven is thus bound to be smaller. On the other hand, this kind of urban town planning has created the need for a personal car as well as the large average distances per vehicle that lead to high levels of total urban pollution in the United States.
- 4 However, no mechanism ensures that this potential technical progress occurs. Whether it does depends on the buyers' preferences, the price of cleaner technology, and the policy instruments used.
- 5 For VOCs and CO, emissions during the first kilometer driven are about half the total emissions of a 100-km trip. The pattern for particulates is probably similar. Fuel consumption and carbon dioxide emissions also increase for the first kilometer driven during cold weather, but not much. The engine typically does not produce higher levels of nitrogen oxides on cold starts; the reason for the emissions increase is that the catalytic converter does not work properly when cold (Holman, Wade, and Fergusson 1993).
- 6 Johansson-Stenman (2002) quotes Krawack (1993), who states that a reduction from 40 km/h to 20 km/h doubles the emissions of carbon monoxide (CO) and volatile organic compounds (VOCs) for a passenger car equipped with a catalytic converter. Because VOCs and CO are 250% higher under congested conditions than during free-flowing traffic, congestion and air quality are directly correlated.
- 7 As the price of this technology falls, the possibilities increase, but there is always a trade-off between allocative efficiency gains and system costs (for a derivation of such second-best regulations, see Verhoef, Nijkamp, and Rietveld 1995).
- 8 Tradable permit schemes would fulfill an analogous role. In such schemes, motorists would be required to acquire additional permits per mile for more-polluting cars and additional (or separate) permits for sensitive zones. Much of the discussion of the fee scheme also would apply to this kind of permit scheme.
- 9 Examples include making electric motor heaters mandatory in cold climates as well as providing the necessary electricity at parking meters and so forth. In the long term, new developments in technology may make such external electric preheating unnecessary.
- 10 Ideally, congestion fees should be charged on congested roads, not new roads. Optimal road pricing must have a substantial steering impact on the magnitude and composition of traffic. A system designed to raise revenues without affecting traffic (as is commonly proposed) wastes resources. In the United States, the road lobby seems to embrace the toll concept to continue building roads as public funds become less plentiful. In Chile, the number of new toll roads increased quickly during the last decade; tolls are used to finance road building. In 2000, 11

total toll roads yielded combined revenues of more than \$100 million, or 20% of the highway department's budget (see Goldmann 2002). Currently, several fully automated tolls exist.

- 11 This wise feature would not currently be possible within the European Union because the Directive on Road Tolls (1999/62/EG) allows only member countries to charge for highways. This regulation encourages diversion into secondary roads where the wear and tear caused by heavy vehicles would require expensive maintenance.
- 12 The effectiveness of road pricing (or of a system of environmental classes) may be scale-dependent. Some of the benefits (such as technology development) would be limited if implemented only in one country, whereas many of the costs (per vehicle) would be much larger. On the other hand, the problems (as well as the tax base) are highly local, and local politicians and motorists are likely to want the proceeds used locally, which would appear to favor some form of fiscal federalism.
- 13 Parts of this chapter build on earlier work joint with Olof Johansson-Stenman.

Taxation or Regulation for Fuel Efficiency

This chapter deals with fuel taxation and its counterpart: regulation of fuel efficiency. A degree of fatalism sometimes creeps into discussions concerning environmental policymaking. Those who suggest drastic taxes are seen as unrealistic idealists. However, the mere range of fuel taxes that exists today (in the developed as well as the developing countries) is amazing. In some countries, fuels are still essentially subsidized, and in others, the level of taxation has raised the effective market price to three or four times the world market price. Although the main motive is not always environmental, this diversity should help illustrate the fact that the range of possible policies is somewhat wider than what everyday policy debates assume.

In this chapter, we not only discuss the importance of prices and taxes for consumption levels but also touch on the political economy of fuel taxes—what it is that makes fuel taxes so much easier in some countries than in others—and how effective the alternatives are in terms of fuel efficiency regulation.

Fuel Taxation

Fuel taxes are not generally imposed for environmental reasons, and in fact, they are not ideal instruments for addressing the complex set of environmental problems associated with road transportation. However, they are almost optimal for global climate issues, although the tax should be on fossil fuels only and not restricted to the transportation sector. Because fuel taxes have varied a great deal between countries and over time, good empirical evidence shows how well they work. The intent need not be environmental for the taxes to be environmentally effective.

Domestic fuel prices vary considerably, even among the developed market economies, primarily because of differences in tax rates¹. Fuel prices in the United States are sometimes as low as one quarter of those in several European countries. Prices in Canada and Australia also are lower than in Europe. Within

Europe, differences in fuel taxes and prices among countries have been considerable, such as between Italy and Germany; during the 1990s, these differences were largely eliminated. Luxembourg, however, remains an extreme case. Its low tax rates attract customers from neighboring E.U. countries, and as a result, Luxembourg reaps big tax receipts (hardly a good example of European solidarity). In developing countries, variations over time are considerably larger, among both countries and petroleum products.

The oil-producing countries in the developing world tend to subsidize domestic consumption (Stern 1989; Stern and Belhaj 1989). Mexico, Nigeria, Saudi Arabia, and Venezuela have low fuel prices, whereas some of their importing neighbors tax fuels heavily, partly in an effort to reduce imports. In addition, many of these governments use heavy cross subsidies to favor different consumer groups; the favored fuel might be gasoline (to please the urban middle class), kerosene (rural poor), diesel for buses (urban poor) or tractors (farmers), or heavy fuel oil (industrialists). Gasoline, kerosene, diesel, and fuel oil are joint products from the same crude oil. The difference in taxation of the many distillates illustrates the politics of distributional effects (see Chapter 12). Complications arise with the pricing of close substitutes such as alcohol. The biggest difference is generally found in poor countries or countries with large, poor rural populations, such as Bolivia, Guatemala, Haiti, Jamaica, Mexico, Nicaragua, and Peru. In Jamaica, during the radical Manley regime, the price of kerosene was 20% of that of gasoline. These policies could not continue in the long term because they led to large-scale adulteration of gasoline with kerosene². One of the few countries in Latin America where no cross subsidization appears to take place is Chile, where market principles are engrained in policymaking.

Another rationale for these tax differentials is based on the notion that intermediate goods should not bear general revenue taxes; however, they should bear taxes intended to internalize externalities (see Chapter 5). If it is administratively impossible to distinguish between fuel for commercial use and fuel for private use, then policymakers may believe that diesel is generally used for commercial transportation (e.g., buses, trucks, and tractors), whereas gasoline is used primarily for private vehicles and that a tax differential between the two may approximate the idea of not taxing intermediate goods.

Variation in product prices is even greater in Africa, where large-scale smuggling has taken place (Stern and Belhaj 1989). Some countries of the Sahel subsidized different fuels in the 1980s to help low-income people and to reduce the risk of fuelwood collection in a fragile ecological environment, but the policies were not coordinated between countries. Senegal chose to subsidize one fuel—liquefied petroleum gas (LPG) in small 2.7-kilogram bottles, intended for household use—to avoid adulteration and industrial use. This policy caused considerable smuggling from Senegal to neighboring Gambia, where the same LPG cost more than six times more. At the same time, gasoline was being smuggled into Senegal from Guinea and other countries that had subsidized gasoline instead. Substantial “trade flows” (of contraband) in opposite directions were created between

neighboring countries in response to differences in subsidies and tax policies. The price reforms that accompanied structural adjustment programs and other developments during the 1980s and 1990s have generally reduced or abolished subsidies. However, most African countries still try to shield their consumers from price and exchange rate variations through subsidies (Wright 1996).

Even in OECD countries, taxes differ considerably. European domestic fuel prices vary considerably among countries, primarily due to differences in tax rates; though there is a minimum fuel tax level mandated by the EU, fuel fiscal measures are determined at a national level and hence several fuel regimes among EU members (Ryan et al. 2009). Diesel is usually much cheaper than gasoline in many countries. There may be several reasons for this. At one time, diesel was preferred over gasoline because it is generally more energy efficient and was believed to create less-toxic exhaust emissions; the role of particulate matter (PM) and the dangers of diesel to human health are fairly recent observations. Therefore, until catalytic converters reduced the emissions from gasoline engines, diesel fuel was favored. At the time, energy efficiency had a greater weight than health issues in policymaking, and diesel was more efficient (also an advantage from the viewpoint of global warming). Today, concerns over the health effects of PM have increased, and more attention probably will be paid to diesel³. Diesel fuel is used by taxis, buses, and tractors and thus affects politically sensitive interest groups. In addition, any diesel policy must address the substitutability of diesel and fuel oil (see Chapter 18).

Effect of Fuel Taxes and Prices on Consumption

It is not uncommon for people to think that fuel taxes will have little or no effect (aside from the perverse effects such as smuggling) because they view transportation as being necessary and relatively inflexible. There is some truth to this belief in the very short run, but on the whole, it is a prime example of common sense leading the noneconomist astray. At the individual level, it might seem that people must make the same journey to and from their jobs, irrespective of the fuel price. At the social level, in the long view, many mechanisms of adaptation exist: people can buy more fuel-efficient cars, carpool, take public transportation, or move closer to their work; such adaptation explains a large fraction of the long-run fuel demand elasticity (Johansson and Schipper 1997). At the level of the economy, the direction of technical progress may be affected as more fuel-efficient cars are produced; even the architecture of towns and societies, public transportation, telecommunication systems, and cultural patterns of interaction may change. Sterner (2007) demonstrates just what an effective policy fuel taxation is. Fuel demand has been cut so dramatically in the EU, Japan, and elsewhere that it actually has had a large and significant effect on the carbon content of the atmosphere—which is more than can be said for any other instrument, including the EU emissions trading system (ETS).

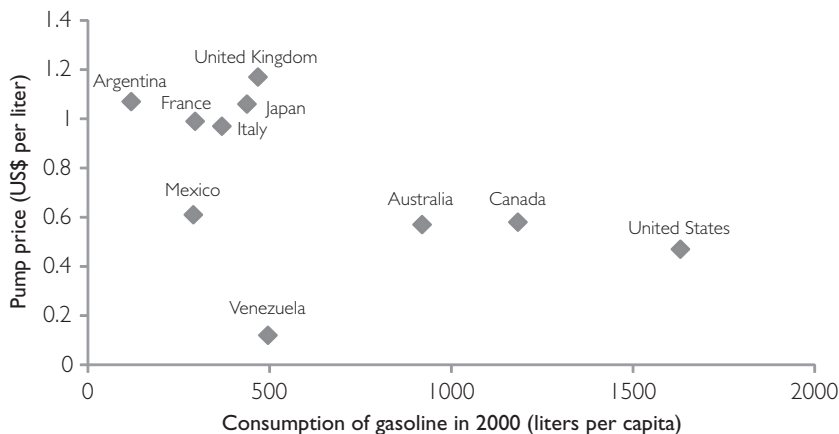


FIGURE 17-1 Domestic Gasoline Consumption Levels and Prices in Some OECD and Latin American Countries

Many of these effects are embodied in or complementary to other investments and become apparent over time, so the final result is complex and best studied by econometrics. A tax increase in one year most likely will not be reflected in a decline in consumption the same year or even the next year. During any given period of time, many conditions change in the economy. The results of policy changes occur in the long run and are difficult to observe because an observed trend must be compared with an unobserved counterfactual situation. A comparison of countries that have had stable but different policies for a long time can reveal some direct effects. For example, countries with cheap fuel (such as the United States and Canada) have much higher consumption rates than many of the European countries, where fuel is more expensive (see Figure 17-1). Similarly, in Latin America, cheap fuel in Mexico and Venezuela leads to higher consumption than in Argentina or Brazil (Rogat and Sterner 1998). In addition, many other factors that determine gasoline consumption cannot be captured in a single diagram but require econometric analysis.

In studies of gasoline demand, surveys show that the choice of model, data set, and estimator does influence results (see Box 17-1)⁴. Generally, short-run elasticities are low, but long-run demand is responsive to income and fuel price (with elasticities not far from unity). In addition, some factors are country-specific, such as the rapid decline in the use of gasoline due to the motor alcohol program based on domestic sugar production in Brazil.

Political Economy of Fuel Taxation

Although every country has different characteristics, gasoline demand is generally sensitive to price—and thus to taxation. This sensitivity is not

BOX 17-1 Models and Data Handling for Estimation of Fuel Demand Elasticities

To select policies, it is necessary to understand policy response, which is often summarized in elasticities. However, measuring elasticities is not a trivial task. Results vary substantially with the model chosen. The simplest model of fuel demand is the static model, M_0 :

$$G_{it} = c + \alpha P_{it} + \beta Y_{it} + \mu_{it} \quad (17-1)$$

where G is gas consumption, P is price, and Y is income; indices i and t refer to countries and years, respectively; c , α , and β are parameters; and μ is an error term.

However, static models do not capture dynamic adaptation, at least not with time-series data. Demand for fuel is derived from transportation demand and has at least two components of adjustment: vehicle use and vehicle stock. Vehicles are long-lived, and adjustments take place over time. Furthermore, the complements and substitutes to vehicles include the entire transportation infrastructure (roads, town planning, etc.), which has an extremely long life.

The adjustment can be modeled either by including all the relevant variables (such as vehicles) or by using dynamic models. An early but widely used representation of dynamic behavior is the partial adjustment model, M_1 :

$$G_{it} = c + \alpha P_{it} + \beta Y_{it} + \eta G_{it-1} + \mu_{it} \quad (17-2)$$

This model assumes inertia in adaptation, leading to the inclusion of a lagged dependent variable, G_{it-1} . The lagged endogenous model assumes geometrically decreasing lags. Less restrictive models are available (e.g., the inverted “V” lag model, $G_{it} = c + \alpha P_{it} + \beta Y_{it} + \eta G_{it-1} + gP_{it-1} + dY_{it-1} + \mu_{it}$, and the polynomial distributed lag model, $G_{it} = c + \sum_t \alpha_t P_{it-t} + \sum_t \beta_t Y_{it-t} + \mu_{it}$, with appropriate restrictions on α_t and β_t where τ is the number of years lagged).

Among data strategies, one possibility is the *time-series approach*, in which each country is analyzed separately. It highlights country specifics but does not fully use the country-to-country information. Another strategy is *pure cross section*, which gives separate elasticities for each year. The most efficient method must be combined *cross section time-series* (Baltagi and Griffin 1983). The choice of model must take into account the character of data used (Pesaran and Smith 1995).

Additional approaches can be used to summarize all the information in one elasticity estimate: pooling estimators, for example, with country dummies (which is referred to as “within”); taking mean group estimates, which are averages of individual country (time-series) estimates or cross section estimates of averages over all the years of the data (referred to as “between”); or aggregating all the country data and performing a time-series analysis on the aggregate.

Notes: For the partial adjustment model, assume that equilibrium gas consumption (G^*) is $G^*_t = c + \alpha P_t + \beta Y_t + \mu_{t1}$ and partial adaptation $G_t - G_{t-1} = s(G^*_t - G_{t-1}) + \mu_{t2}$ (where s is the rate of adaptation). For more information about the Koyck transformation, see any econometrics textbook (e.g., Gujarati 1988).

great in the short run but is in the long run. Fuel taxation can be a powerful instrument to limit the use of fuel but cannot be used without considering the availability and prices of substitutes such as public transportation. It is

BOX 17-2 Fuel Taxes or Permits: A Politically Possible Instrument for Global Warming?

As part of a very crude policy experiment, assume that fuel demand is determined by $G = YP^{-0.8}$, where Y is income, and P is price (as suggested by the results in the section “Effect of Fuel Taxes and Prices on Consumption”) and that these elasticities are valid over a large interval. Assume also that a 50% increase in income is predicted and a 50% decrease in carbon emissions desired (a long-term goal for the whole economy suggested by the Intergovernmental Panel on Climate Change; these values are for the transportation sector, but similar policies would be applicable to other sectors).

In this case, the suitable market price could be calculated from the formula $P = (G/Y)^{-1.25}$, and the price needed would be four times higher: $4 = (0.5/1.5)^{-1.25}$. In principle, such a price hike could be achieved through either taxes or tradable permits. To many people in countries with cheap fuel, it may seem impossible. However, many countries (primarily in Europe, but also in oil-importing countries in the developing world) already have shown that it is indeed possible.

not necessarily easy to pursue an independent tax policy without considering the policies in neighboring countries. Finally, fuel taxes are not an optimal instrument for most transportation-related externalities. If they were levied on the fossil content of the fuel, they would be a correct instrument for global warming (see Box 17-2). They still would not be an efficient instrument to regulate local pollutants, however, because they would do nothing to encourage technical abatement such as improvements in motor performance to reduce exhaust emissions.

The “solution” of Box 17-2 is conceptually simple but politically difficult. Why are fuel taxes not more frequently and uniformly used in OECD countries? Most countries declare an intention to reduce emissions, and most economists would agree that the most efficient way to do so is to increase taxes. Taxes are not designed to fulfill only one goal; in fact, they often are not “designed” at all, at least not in an abstract sense. Instead, they evolve through political processes in which various political stakeholders fight.

Most fuel demand studies assume that causality runs from taxes and prices to demand. Suppose that the causality runs the other way; the motorists (and voters) in countries with historically high levels of fuel consumption may be most adamantly opposed to tax increases, whereas those in countries with low consumption might find it easier to tolerate them. Gasoline taxation was examined in several OECD countries using Granger noncausality tests; the analysis confirmed that in addition to the demand relationship, the existence of a separate causal relation from quantities back to prices cannot be ruled out in many OECD countries (Hammar et al. 2004)⁵. This finding supports the notion that the political momentum built up in support of one institutional arrangement may influence the choice of policy instrument. Flood et al. (2010) show, however, that this finding does not significantly affect the demand elasticities estimated.

Models of vote maximization (e.g., Hettich and Winer 1988) assume that successful politicians will avoid overtaking their voters. However, governments need money to operate, and something must be taxed—income, wealth, property, or consumption goods. Consumers may resist any tax, and fuel taxes more so if those who bear the greatest burden have more political power than other groups. Fuel taxes also might be resisted less if they are seen as doing some environmental good or benefiting groups with limited influence (or groups with a high ability to pay).

The political attitudes toward mobility and environmental pollution may be decisive, and the costs and benefits depend on population density and other factors. An analysis of interstate differences in fuel tax within the United States presents the following observations (Goel and Nelson 1999):

- 1 The presence of an oil industry leads to lower gas taxes. The interpretation of this statement in terms of vested interests, employment, and lobbying seems straightforward. Similar mechanisms appear to apply to Mexico, Venezuela, and the countries of the Persian Gulf (but not to Norway or the United Kingdom), leading to disproportionately low taxes and high consumption levels.
- 2 Higher highway tolls are associated with lower taxes. The tolls are presumably seen as alternatives to fuel taxes for managing demand or financing highways.
- 3 Higher population density appears to affect the level of taxes in two opposite ways. Population density increases health costs while reducing the costs of building highways and providing public transportation. Low population density makes the automobile a necessity, leading to lower taxes; whereas high population density reinforces the externalities, leading to higher taxes.
- 4 Higher compliance with environmental standards means higher taxes. This statement appears to be logical (although causality could go in either direction).
- 5 Nominal rates tend to be adjusted to inflation, and higher real (pre-tax) prices of gasoline lead to lower taxes. These statements suggest that politicians tend to seize the opportunity to raise taxes whenever it is relatively easy, for instance, when (pre-tax) gas prices decline.

The causality between political economy and fuel taxation could result from several scenarios. In high-consumption countries such as the United States, consumers own vehicles and property, and their lifestyle hinges on a high use of fuel. Thus, they stand to lose a lot of money from any fuel tax. Many businesses—car producers, gas stations, oil companies, and even amusement parks and shopping centers—have vested interests in keeping the cost of gas low. Employees of these businesses share this interest, to the extent that their jobs depend on the profits of their employers. U.S. oil companies, a notoriously

powerful lobby, oppose fuel taxes. Therefore, the political representatives of all these parties have a lot of popularity to gain in making the case against fuel taxes. At the same time, the parties who have something to gain from higher taxes are either few or diffuse and unorganized. On the one hand, providers of alternative forms of transportation—conceivably, those employed by or with interests in public transportation, bicycles, and so forth—might gain from higher fuel taxes. On the other hand, in principle, the general public might gain from a better tax system and the resulting improvement in allocation within the economy, but this abstract concept is not likely to attract much support.

In a country like Italy, where little oil is produced, the balance of interests might be somewhat different. A reasonable strategy to keep oil imports down is to conserve energy by raising its price. The fact that this strategy happens to derail some of the resource rent from foreign producers to the national treasury can hardly be a problem. Most motorists have small vehicles and live close to their jobs. Fuel prices have been high for so long that motorists have adapted, so they do not have much to lose from even higher taxes. Furthermore, the owners, employees, and subcontractors of Fiat know that a high market share of its small, fuel-efficient automobiles depends on high fuel prices. The employees of the public transportation sector also benefit. The fact that income taxes have proven notoriously difficult to collect in Italy may be the real pragmatic reason behind the high gasoline taxes, which are considered an easy source of tax revenue. Thus, Italians who feel that the government needs its revenues—because they appreciate the services it provides or because they are government employees—may implicitly support the fuel taxes⁶.

The tradition of earmarking fuel taxes also differs across countries and may influence the support for or opposition against such a tax. For example, in the United States, most fuel taxes are earmarked for highway construction. This practice may appear odd to economists used to preaching the virtues of not earmarking but has general political support in the United States.

At a general level, cost distribution is bound to be an important factor for the politics of industrial policy. In the United States, the absence of alternative means of transportation coupled with an uneven income distribution make fuel taxation particularly sensitive for low-income groups. In many European countries, public transportation offers reasonable alternatives and shelters low-income households somewhat from the cost of higher gas prices. In countries with a more even distribution of income, gas price would not be such a major concern. In poor countries, where automobiles are a luxury item, a fuel tax should be progressive. However, the affected elite often have such power that they can block the implementation of such taxes.

The argument is commonly made that if fuel (particularly diesel) prices rise, then all other prices in the economy will rise as a result of the increased cost of transportation. To an economist, this argument does not appear to be strong because the cost share of transportation services is typically small in the final price of most goods. Also, if fuel prices rise as a result of taxes, then these taxes

may be assumed to replace other taxes (e.g., on labor, which also would have raised some prices). A large share of the resistance against fuel taxes often comes from professional truckers and farmers. For instance, Decker and Wohar (2006) find that the freight trucking industry's contribution to total state employment is a highly significant determinant of a state's diesel tax rate in United States, consistently suggesting that the greater this contribution, the lower the tax rate.

As already discussed in Chapter 12, the argument is often made that fuel taxes are regressive and hurt the poor. According to some 25 case studies of different countries in Sterner (2011), this is at least not true in low-income countries, where the fuel tax tends to be quite strongly progressive. For high-income countries there may be some weak regressivity but this can easily be remediated (if so desired) by using the proceeds of the tax to lower other more regressive taxes. For commercial vehicles, driving is not a consumable good but an intermediate good (i.e., service) and thus should not be subject to taxes intended to raise revenues (see Chapter 11). However, nonrevenue motives such as internalizing external costs are not affected (optimal road tolls under distortionary taxation are discussed in Calthrop and Proost [1998] and Calthrop et al. [2000]). The actual taxation rate and price of the fuels do not necessarily have to differ at the pump. The differentiation can (at least in principle) be achieved by tax deductibility (for the revenue-raising taxes) in the accounts of the firms that use the fuel for commercial purposes. Such an approach may help to reduce the political tension surrounding fuel taxes.

Another important determinant of fuel consumption is population density. Few international studies of fuel demand include this variable, so it is difficult to show the connection empirically. It may be because population density is defined nationally, whereas the most important determinant is local population density. However, such a variable is difficult to construct because it is partly endogenous. Most U.S. cities have population densities much lower than those in Europe (in fact, lower than many "rural" areas in developing countries). It is not surprising that fuel consumption in the dispersed U.S. cities, which often lack satisfactory public transportation, is four times as high as in typical European cities. Although a large share of the difference is due to habits and vehicle characteristics that would adapt to changed fuel prices within 5–10 years, another large share is due to differences in urban architecture, which would take considerably longer than 10 years to adapt.

Regulations Instead of Price Mechanisms

Despite all the discussion thus far about market-based instruments, the most usual policies for road transportation are still physical and regulatory: traffic management, speed controls, mandatory equipment (e.g., catalytic converters), fuel quality regulation, mandatory inspection and maintenance, and town planning (see Chapter 18). The policy instrument most commonly

used to reduce vehicle pollution is emissions standards. In a first-best world without control costs, the standards would be superfluous and inefficient because emissions standards give excessively clean cars in the country and insufficiently clean ones in city centers.

In principle, an optimal set of road pricing charges would make it more expensive to drive in the city center, and an optimal mix of vehicles would result without implementing emissions standards. In reality, several factors make it unwise to abandon emissions standards. Because of the economies of scale in car production, the high costs of monitoring, and uncertainties about the actual effects of road pricing, keeping some form of conventional emissions standards seems prudent. However, with road pricing, the role and character of the compulsory emissions standards would presumably change⁷. Their importance for pushing technology ahead would diminish, and their role might be more of a safe minimum standard applicable to general conditions. The main driving forces for cleaner vehicles would be reductions in environmental road charges.

Another policy is zoning. Swedish municipalities are allowed to declare certain urban zones “sensitive” and ban certain kinds of vehicles within these zones. Typically, environmental vehicle classifications are used. Swedish authorities continue to lobby in favor of environmental classes, and these classes may be used by local government in selecting goods for public procurement, for instance (Naturvårdsverket 2002).

CAFE in the United States

Surprisingly, the country generally thought of as the foremost supporter of free markets has relied almost exclusively on direct regulation of fuel efficiency instead of using fuel taxes. In the early 1970s, reducing fuel consumption was a universally accepted goal in the United States, not for environmental reasons but for reasons of national security. Hence, it would appear that the United States tried to choose effective policy instruments when it mandated the Corporate Average Fuel Economy (CAFE) standards in 1975 as the main instrument for influencing transportation fuel demand. The CAFE standards required all manufacturers of motor vehicles sold in the United States to meet a fleet-average fuel economy of 18 miles per gallon (mpg) by 1978. The standard was intended to increase the fleet-average fuel economy to 27.5 mpg by 1985, but by then, fuel prices had dropped, and Ford and General Motors managed to lobby the U.S. Department of Transportation to lower the standard to 26.0 mpg temporarily. From 1990 until 2010, the standard has been 27.5 mpg. The latest CAFE standard, released in 2010, is intended to reach a fuel economy of 34.1 mpg beginning with the 2012 model year and increasing in stringency through model year 2016. Moreover, the official projections are that CAFE standards will entail by 2016 an improvement of nearly 100% over 1978 cars; it is estimated that over the lifetime of the cars of 2012–2016 make,

subject to the latest CAFE standards, the fuel savings will amount to 61 billion gallons and an equivalent of 650 million metric tonnes of CO₂.

A number of studies have considered fuel economy standards in the context of comparing alternative policy instruments. For instance, Kleit (2004) and Austin and Dinan (2005) show that the welfare costs of CAFE standards were much higher than if a corresponding fuel tax had been used; some studies even doubt that the standards had any aggregate fuel-saving effect at all. Kleit's estimates indicate that during the period of study, the costs of fuel savings through increased CAFE standards was US\$1.09/gallon saved. Instead, the same cost savings could have been achieved through an increase of US\$0.11/gallon in the gasoline tax. The advantages of an increased gasoline tax over higher CAFE standards are also shown by Austin and Dinan (2005). They compare the cost of tighter CAFE standards over a 14-year period with the cost of a gasoline tax designed to save the same amount of gasoline over that period and conclude that a gasoline tax would save the same present discounted quantity of gasoline at a cost that is 71% lower than the comparable CAFE policy. On the other hand, Parry et al. (2007) divide automobile-related externalities into those arising from gasoline use and those from miles driven, showing that even if CAFE standards reduce fuel consumption, gasoline taxes reduce a greater number of important externalities than do CAFE standards. For example, CAFE do not directly address automobile accidents or congestion.

Several phenomena reflect the inefficiency of CAFE as a policy instrument. In a classic example of the market finding its way around physical regulations, minivans, light trucks, and sport utility vehicles (SUVs) have become increasingly popular, and their sales as personal vehicles have increased significantly. (Minivans, light trucks, and SUVs are not covered under the same stringent regulations as cars.) From 1980 to 1993, the number of passenger cars sold in the United States fell from 8.98 million to 8.52 million per year, whereas the number of minivans, light trucks, and SUVs sold more than doubled from 2.2 million to 5.0 million per year.

The level of penalty was relatively modest (US\$5/vehicle for every 0.1 mpg shortfall)⁸. Several European manufacturers simply paid the fines, whereas domestic producers had access to other mechanisms. U.S. companies usually did not even pay the fairly modest fines. For several years, Ford and General Motors succeeded in their lobbying to roll back the standards. Another allowable strategy was to use "banked" credits from earlier overcompliance (i.e., scoring under the average in some earlier years) to compensate for penalties. Credits also could be gained through the sales of special alternative-fuel vehicles, and the companies were able to avoid legislation by moving production abroad (or back to the United States) because the quotas for locally produced cars and "imported" cars—which included U.S. companies that manufactured vehicles abroad—were different⁹.

It is instructive to consider why the CAFE standards were chosen as a policy instrument. The simple answer is that raising taxes was politically

infeasible. Various administrations have unsuccessfully tried to raise gas taxes in the United States; for instance, the Clinton administration managed to approve only a small increase of US\$0.043/gallon after proposing a significantly larger increase first.

The situation appears to be a clear case of the political power of lobbies, but why are they so successful? Models of political influence find that the strongest cases are when the benefits are enjoyed by a few and costs are spread widely. Although the allocative costs of CAFE regulations are spread widely, so are the benefits of low gas prices. However, effective lobby organizations seem to take care of organizing the motorists. Another explanation is that Americans attach little value to added government revenue. In that case, people may see the gas tax as a loss (much like the effect of rising international prices), particularly if they do not believe that other taxes will be reduced or that increased government spending will increase welfare. This aspect seems to be at least one important part of the explanation, because the past two decades have been a period of intense political opposition to “big government” and taxation in the United States. In one study, motorists in southern California were somewhat more amenable to paying for congestion fees if part of the fees were refunded to motorists (Harrington et al. 2001).

Other reasons why CAFE standards were chosen are related to the political economy of gasoline taxation: Americans have become dependent on cars because of inadequate public transportation, the culture of mobility, and low population density—even in towns and cities, where “urban sprawl” is becoming more and more commonplace. In turn, population density is low and public transportation inadequate because of the large number of cars and the prevalence of low gas prices for almost a century. In addition to lobbying by industry and motorists, in several states (e.g., Wyoming and New Mexico), congressional representatives will fight fuel taxes for fear that their states will lose tourism revenues or that their constituents will be hurt by higher costs of commuting. The fact that the car is a means of mass transportation in the United States, more than anywhere else in the world, also suggests that income distribution concerns may weigh more heavily. In fact, one study finds that gas taxes are mildly regressive in the United States, although the shares of income are small and vary little except at the ends of the distribution, i.e., the top and bottom deciles (Poterba 1991)¹⁰.

Supplemental Reading

Fischer et al. 2007a,b
Harrington 1997
Hughes et al. 2008
Lyon et al. 1995

Pock 2010
Portney et al. 2003
Small and Kazimi 1995
West and Williams 2005

Notes

- 1 The relationship between taxes and domestic prices is complex, and there are many reasons for the differences in domestic fuel prices between countries. Part of the price at the pump is the cost of service, refinery, fuel transportation, and so forth. These costs are determined by geography, scale, efficiency, cost levels, and company and income taxation in each country. Generally, however, they are a small part of the total price, and the main reason for variations in domestic prices is differences in excise tax. I focus primarily on gasoline prices because the data tend to be clearer for gasoline than for diesel, where the substitution for fuel oil causes some problems.
- 2 Adulteration and smuggling are not restricted to developing countries. In 1998, about 12.5 million liters of gasoline were adulterated with thinner to avoid taxes in Sweden.
- 3 Knowledge is always incomplete, and ironically, some signs indicate that the preference for gasoline over diesel may be reversed once more. New filters and other technologies reduce the particle emissions from diesels, and Otto engines (which use gasoline) reportedly produce more of the finest (nanometer-sized) particles, which may have worse effects on human health than larger particles.
- 4 Dahl and Sterner (1991a, 1991b) present a survey that stratifies results by model and data. In subsequent work, Sterner (1991), Sterner and Dahl (1992), Sterner and Franzén (1995), and Sterner, Dahl, and Franzén (1992) estimate different models for one consistent data set from OECD for 1960–1985. Rogat and Sterner (1998) estimate the same models for some Latin American countries.
- 5 In complicated systems, many causal relationships may overlap (e.g., prices influence demand, and demand, in turn, influences tax policies and thereby prices through political pressures). The Granger test of noncausality is one statistical method of differentiating these relationships.
- 6 Ironically, even the oil companies may benefit. In high-tax countries such as Norway, Sweden, Italy, and most notably Japan, the high taxes are combined with high pre-tax prices of gasoline. This combination seems odd (higher taxes should squeeze the margins of the oil companies) but may have a political explanation in the implicit acceptance of higher profits in exchange for high taxes. One can imagine the authorities desiring a high price level to promote conservation and “sharing” the rent with the gas companies.
- 7 This is not the same as mandatory technology such as the catalytic converter. In principle, it is more efficient if companies are free to test new technologies intended to meet a specified goal.
- 8 This penalty applied only to sales of more than 10,000 vehicles/year in the United States. Even a dramatic shortfall of 5 mpg would warrant a fine of only US\$250—less than 1% of the price of a new car, or the cost of the extra fuel for just one year of driving (assuming 25,000 miles/year and US\$1/gallon).
- 9 Ford used this strategy at least twice, moving the production of rear axles to Mexico and then back again for “CAFE-related” reasons (Thorpe 1997).
- 10 Parts of this chapter are based on joint work with Åsa Löfgren and Henrik Hammar.

Fuel Quality Policies

Chapter 17 focused on policies for fuel efficiency and for carbon emissions. In this chapter, we turn to fuel quality, a policy instrument frequently used to deal with regional and local environmental problems caused by transportation. In view of climate change and the increasing prices of fossil fuels, biofuels have taken center stage in policy debates during the last decades. At the end of the chapter, we discuss the opportunities, but also the limits, of the use of biofuels for transportation.

Fuel Quality and the Phaseout of Lead

In addition to vehicle design and traffic management, fuel quality is an integral factor in formulating policy for the transportation sector.¹ The use of lead in gasoline has become a symbol of the health effects caused by transportation. Originally, health effects were among the indirect reasons for phasing out leaded gasoline; the main reason was that catalytic converters become “poisoned” (i.e., they cease to operate) when contaminated by even small amounts of lead.

Catalytic converters were introduced for environmental reasons, but of a different kind. The original main concerns were smog-related emissions—nitrogen oxides (NO_x), volatile organic compounds (VOCs), and carbon monoxide (CO)—in sunny climates such as in California. However, as knowledge of the health effects of lead exposure increased, it became clear that exposure to lead can cause loss of intelligence, nervous disorders, hypertension, and even life-threatening consequences in cases of high exposure (CDC 1991; Hayes et al. 1994; Hu et al. 1996; Kim et al. 1996; Lovei and Levy 1997; Needleman et al. 1979; Schwartz 1994; WHO 1987). Children are at greatest risk, mainly because their nervous systems are developing. Malnutrition, particularly the lack of essential trace metals, appears to increase the negative effects of lead exposure, making it particularly dangerous for the poor.

Lead (usually tetraethyl lead) used to be routinely added to gasoline in most of the world because it was a cheap source of increased octane and a good valve lubricant. As unleaded gasoline was becoming the baseline technology, the

production and distribution of unleaded gasoline became less costly than that of leaded. A combination of refinery reconfigurations and nonrefinery innovations further reduced total phaseout costs (Hilton 2006). For example, an individual refinery may refurbish machinery and extend some of its processes, change its crude oil mix, or buy various additives or components such as alcohols or ethers. At the national level, additional options include liberalizing imports, allowing foreign companies to set up refineries, tightening fuel specifications, and possibly implementing incentive-based policies. In some countries, the relevant authorities introduced several grades of gasoline; where only one grade is available, it must be a fairly high-octane gasoline designed to suit the most demanding vehicles, even though the majority of vehicles do not need such formulations. The availability of several grades can thus lower the total demand for octane boosters.

Policy Instruments for Eliminating Lead from Gasoline

Before considering policy design, the “optimal” pollution level must be determined for each source. The desirability of simply eliminating lead exposure as much as possible is universally accepted in developed countries, but this issue might not be so clear in countries where people have very low incomes—particularly because they must carefully decide how to allocate scarce resources between various sources of lead poisoning.

Despite the extensive use of lead in gasoline, gasoline was not and is not a major source of lead in the environment; however, vehicles are the largest source of atmospheric lead in many cities and account for up to 90% of all lead emissions into the atmosphere (Brunekreef 1986). Because lead in this form is easily accessible, such emissions account for a large share of exposure and health effects for many people. In industrialized countries, many uses of lead are banned or heavily regulated². In developing countries, water pipes, glazing, and other sources close to the food chain typically account for a large share of exposure and should be the top priority, although lead in fuel is also important.

No single policy instrument is optimal for phasing out leaded gasoline because the character of technology and markets vary considerably over time and between countries. The set of experiences currently available point to certain advantages and disadvantages of each policy instrument in different settings. The U.S. phaseout of leaded gasoline in the 1970s was different from a phaseout in El Salvador today. Knowledge of the alternatives, refinery technology, effects on motors, and so forth has increased. In addition, the situation in a large country with complex markets and many refineries that have individual interests is very different from that in a country with only one state-run refinery. Approaches used thus far include tradable lead permits, direct regulation, preferential tax treatment, eco-labeling, and lawsuits.

The United States was the first country to phase out leaded gasoline. Some of the background factors (e.g., smog in California and catalytic converters)

have already been discussed in many previous chapters. In 1973, the U.S. Environmental Protection Agency (U.S. EPA) had already succeeded in drastically reducing the lead content of gasoline. In 1982, U.S. EPA wanted to phase out the remaining lead as quickly as possible, which turned out to be just over five years (starting in 1982)³. Although the total benefits outweighed the costs considerably, refineries worried about meeting the fairly rapid pace of the phaseout showed serious resistance (Tietenberg 1994). Producing high-quality unleaded fuel requires appropriate technology, which can be expensive for some refineries, depending on their age and size. For a refinery in a process of expansion and modernization, the additional cost of complying with new regulations for removing lead may be almost insignificant. But for a refinery at a different stage in the investment cycle, the additional cost might be devastating unless it can be delayed to coincide with regular overhaul and maintenance. For some kinds of small refineries, the costs might be prohibitive if appropriate small-scale technology is unavailable. In the 1970s, the United States had many small refineries with small output but sizable influence.

Almost any environmental regulation places some firms at a competitive disadvantage vis-à-vis their competitors and causes legitimate complaints by industry. U.S. EPA feared that lead phaseout legislation would be challenged in the courts and that companies would be able to delay it considerably. This expectation was far from hypothetical; up to 80% of U.S. EPA regulations are challenged in court (Davis and Mazurek 1998, 23). Therefore, U.S. EPA sought a mechanism that would allow individual plants some flexibility without delaying aggregate compliance. The mechanism chosen was a kind of tradable emissions permit that targets pollution intensity—in this case, the lead contents of gasoline; it was not intended to affect lead use by affecting gasoline demand itself (see Chapters 6 and 11).

Figure 18-1 illustrates the U.S. lead phaseout at the aggregate level. A credit was earned by a refinery that used less lead than the standard for that quarter and could then be sold to another refinery that used more lead than the standard. In 1984–1985, small refineries (<5,000 barrels/day) typically used at least 1.5 grams lead/gallon. The possibility of buying rights from the larger refineries apparently reduced the compliance costs significantly for the small refineries, and it also provided incentives for more efficient technology adoption decisions (Kerr and Newell 2003). Starting in 1985, U.S. EPA also granted refineries the right to “bank” credits for later use, and this system was used heavily. As a result, larger-than-required reductions in 1985 made it possible to exceed limits somewhat during 1986 and 1987, when the program was terminated. A significant amount of both banking (10.6 billion grams [Hahn and Hester 1989]) and trading took place, and the estimated savings were several hundred million U.S. dollars.

Although the U.S. lead phaseout scheme was successful in accomplishing its goal, it was not necessarily an optimal or efficient policy instrument. However, it did avoid deadlock in the courts and phased out leaded gasoline

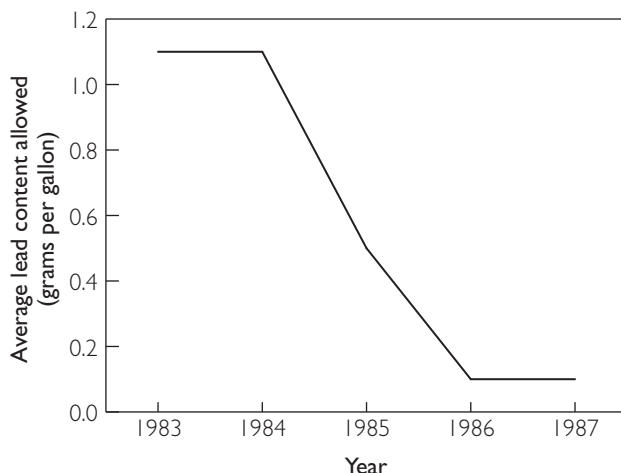


FIGURE 18-1 Phaseout of Leaded Gasoline in the United States

in a timely fashion. One unexpected by-product of the system was rent seekers—companies that were set up to blend fuels and then resell them to claim “lead rights.” They were a nuisance to U.S. EPA and to the refineries that were “cheated” out of these rights. The details of this kind of program have to be worked out carefully; allocating rights in proportion to output—which is essentially what this program did—entails an implicit element of subsidy and risk of rent-seeking (see Chapter 6). These problems would have been avoided if the rules applied only to existing refineries and allowed U.S. EPA to vet new applicants before making them members of this phaseout scheme.

Other countries have not copied this scheme, even though most have phased out or are phasing out lead from gasoline, in many cases a decade or two after the United States. Scandinavia, Germany, and Austria phased out lead rapidly, and the E.U. Council of Ministers banned leaded fuel after 2000 (with some exemptions until 2005), but some European countries lag behind. Japan, like the United States, was early in introducing catalytic converters and phasing out lead in the late 1970s, apparently with no other policy instruments than the “advice” of the Japanese Ministry of International Trade and Industry (quoted in Nivola and Crandall 1995).

In most countries, the main driving forces in addition to the concern for public health were catalytic converters. This fact tends to make the automobile industry an ally. Once a company has converted to producing cars with catalytic converters (which, in some cases, might have been a somewhat reluctant process), it becomes a strong advocate and wants to limit the use of lead, raise its price, and do anything that helps promote the new “clean” cars that it wants to sell. The pace of technological renewal varies among companies, and these differences shape different national policies for the phaseout of leaded gasoline.

Phaseout of Lead in Developing Countries⁴

Prejudice might suggest the phaseout would be slower in poorer countries, but most developing countries have already phased out lead completely: Bangladesh, Haiti, Nicaragua, Vietnam, Albania, Belize, Bolivia, Costa Rica, Dominican Republic, Ecuador, Egypt, El Salvador, and Honduras (Hilton 2006). China's largest cities have also reduced gasoline lead to very low levels (Hao et al. 2006).

Exceptions are the oil-rich countries (Peru, Venezuela, and the Trinidad Antilles, which is a special case), which should have the financial means and know-how to implement a lead phaseout. Ironically, the existence of a strong national oil company trying to resist imported technology could be setting these countries back.

Several eastern European countries made early progress: Slovakia made catalysts mandatory in 1993 and phased out lead in five years. Poland reduced the maximum limit from 0.32 grams/liter to 0.15 grams/liter in 1993, made catalysts mandatory in 1995, and decided to phase out lead in 2000. Bulgaria, Croatia, Hungary, Latvia, Poland, Slovakia, Slovenia, Turkey, and Ukraine (about half of the 18 eastern European countries mentioned in DEPA 1998) are using economic incentives such as lead taxes to facilitate the transition. Most of the older vehicle stock appears to be designed for low-octane fuels, which makes the phaseout easier. The recent imports of (used) "western" cars that need higher octane make it important to phase out lead before demand increases.

The literature on phasing out lead in developing countries teaches several interesting lessons:

- Refineries are complex examples of joint production, making the design of policy instruments tricky. However, if there is a single local monopoly, regulation may be preferable.
- In small countries that have only one refinery, the phaseout can be relatively easy if lead-free fuel or additives can be imported. The experience of El Salvador (less than one year) and other Central American countries shows the advantages of a rapid phaseout, avoiding the cost of dual distribution systems. The introduction of an additional low-octane gasoline may capture a share of the market, allowing the oil companies to save their octane boosters for motorists who really need it, thereby reducing refinery investments.
- Pricing can be an important policy instrument⁵. If unleaded gasoline is more expensive than leaded gasoline, then a phaseout proceeds slowly. Vehicles with catalytic converters risk damage due to intentional or unintentional "misfueling." Policymakers often balk at the political cost of raising the price of leaded gasoline. However, in Jamaica, production cost was lowered by reducing the octane number, such that revenues could be increased without raising the price to consumers.

- In Mexico, a type of labeling was used: Taxis and buses that operated on lead-free fuel were painted green instead of yellow so customers could choose⁶.
- In Azerbaijan, Kazakhstan, and Uzbekistan, gasoline is produced and sold in micro operations, and government control is so weak that lead content cannot be monitored (World Bank 1998a). In fact, lead content may be higher than legal limits. A tax on the lead content of fuels would not be feasible, but a tax (or ban) on imported lead additives might be. Even the low-income Southern Africa Development Community (SADC) countries appear to have phased out most of the leaded gasoline. According to Phiri (2006), a handful of SADC countries—South Africa, Mauritius, Botswana, and Namibia—even beat a 2006 deadline to become lead-free.
- Brazil is a special case with its large ethanol program. Lead is completely phased out despite the country not having catalytic converters on even half the vehicle stock. Forty percent of vehicle fuel is ethanol, and in the rest, ethanol (which has a high octane value) is used as a blending agent.
- Auto and oil companies often support the lead phaseout (at least in the later phases, when they have already been forced to adapt to a ban on lead in domestic markets and find that they can reap the benefits of product standardization by requiring other countries to follow suit); the World Bank has organized conferences and published reports supported by such companies⁷. The World Bank has also provided some technical assistance, but lending to refineries has been a minor component. Its main function appears to be as a catalyst, using its credibility to encourage other institutions to finance the main investments.

Policies for Fuel Quality in Sweden and Other Countries

One of the factors driving the need for the lead phaseout was the harmonization of fuel quality between neighboring countries. Although the health effects of lead are especially dangerous, numerous other additives, impurities, and components (e.g., polyaromatics, sulfur, oxygenates, and phosphorus) also must be controlled. Countries have different fuel requirements that hinder trade in refined products, increasing costs and decreasing opportunities for collaboration and learning. One might not think that fuel requirements would lend themselves to economic instruments, but some examples exist. For example, for some time, Sweden has used environmental classes for gasoline and diesel fuels, providing oil companies the opportunity to be proactive. The cooperative company OK tried green labeling and marketing to get larger market shares by being first to introduce environmentally improved fuels, such as unleaded gasoline. In the early 1990s, Sweden also used strong tax differentiation on fuel oils by environmental class. It encouraged the rapid introduction of cleaner diesel fuel by compensating the oil companies for increased production costs (see Figure 18-2). The oil companies even started

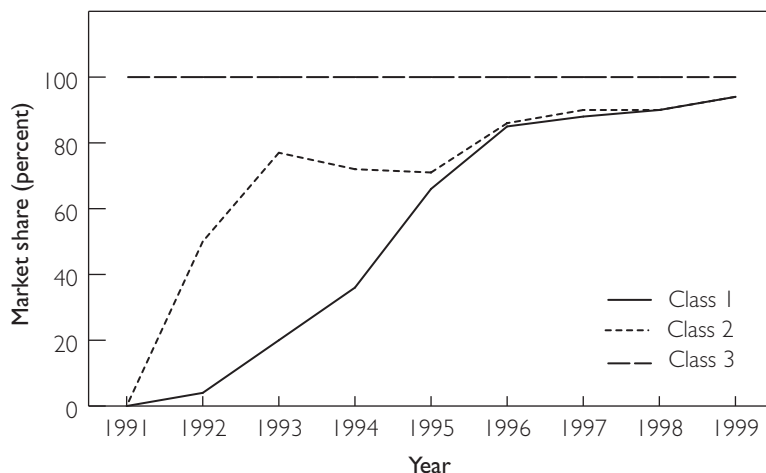


FIGURE 18-2 Market Shares for Environmental Diesel Classes 1 and 2 in Sweden

to sell “environmental” heating oil, capturing 36% of that market in 1993. This result was hardly the intention of policymakers, because environmental advantages are limited outside the transport sector. The subsidy was subsequently removed for this category.

The availability of cleaner diesel had many positive secondary effects. First, it allowed Sweden and Finland to provide an early market for the introduction of particle filters for diesel-powered vehicles. It appears to have been a good example of technology forcing, because early on, there was some doubt as to how these filters would work, and successful demonstration was important for further international marketing. It also allowed environmental authorities to tighten emissions standards for diesel-powered vehicles. Additional secondary benefits were that the combination of cleaner diesel and particle filters made it possible to install better exhaust gas recirculation systems on some buses and trucks, greatly reducing NO_x emissions. Still, various problems have surfaced, including the inability of the current tax law to distinguish between fossil-based diesel and diesel made from biological raw materials, which still has to pay a carbon tax. Another anomaly is that some clean fuels do not qualify for the lower Class 1 taxes because they fail to meet technical specifications such as cetane index or density, which is not central to the environmental issues.

A similar program for gasoline quality defines Environmental Classes 1, 2a, 2b, and 3. This program had somewhat similar effects to the programs for unleaded gasoline and environmental diesel. The introduction of tax rebates to offset increased production costs was used to phase out the more polluting fuel. In this case, a much smaller tax differential (<US\$0.01/liter) was sufficient to cause a virtual takeover by the two improved qualities. Several dozen technical parameters are specified, including the maximum contents of olefins, certain alkanes, and aromatics as well as total and individual oxygenates, such

as alcohols and ethers. Standards for phosphorus, sulfur, and lead are ambitious by international standards and have been tightened over time⁸.

These details have caused some conflicts in which smaller producers of new (sometimes biomass-based) fuels have found that the standards have been designed to suit the large fossil fuel-based companies. It is extremely important that the environmental classes (and all policy instruments) be designed from neutral, general principles related to actual environmental damage. Building them on the solutions or technologies that happen to be available at one point in time is tempting but risky and detrimental to long-run technical progress. The temptation lies in the relative ease. Regulators may otherwise be accused of creating rules that are impossible to abide by or taxes for categories that may never exist. If a concrete technology can be given as an example, regulators feel protected—but run the risk of creating regulations that are overly influenced by a particular brand of technology.

Some inconsistencies are found in the regulation of diesel fuel. Fuels with biological origin still have to pay carbon taxes, and the taxes on different environmental classes cannot claim to be based on estimates of damage. In fact, the relevant tax law is written in such a way that any fuel or fuel ingredient used in a gasoline-powered (i.e., Otto engine) vehicle is taxed the same as gasoline. This ruling even applies to water and, for instance, hydrogen produced from wind power. By reasonable judgment, some fuels produce lower emissions but still do not qualify for the best environmental class. Since Sweden joined the European Union, its ability to decide environmental classes on its own has been limited; Environmental Class 3 (the lowest) has been harmonized with the EU requirements.

With increasing globalization, all countries are exposed to increasing pressure to harmonize standards, regardless of whether they are members of a geographical trading block. This trend has both positive and negative aspects. On the one hand, it may limit countries that wish a faster pace of improvements, but on the other hand, it provides a powerful force for modernization and improvement in environmentally less advanced countries. In many countries, the World Bank and the United Nations (e.g., through their joint Energy Sector Management Assistance Program [ESMAP]) are active in promoting the harmonization of fuel specifications (UNDP/World Bank ESMAP 1998). Another important factor is the advent of powerful and relatively inexpensive technologies for detection of fuel adulteration. Thus, in 2000, even in Kenya, diesel was being sampled in purity inspections that are able to detect even small quantities of kerosene (a popular fuel used for adulteration because it is so much cheaper than diesel).

Biofuels and Transportation

Since the beginning of the automobile era, oil has been the main energy source for transportation. For example, in 2007, petroleum products accounted for

more than 95% of U.S. transportation energy. However, some decades ago, an intensive search for alternatives started, driven by a variety of concerns including high and volatile oil prices, increased demand, uncertainty about petroleum supplies, reliance on imports from politically unstable regions, and greater recognition of the environmental damages caused by fossil fuels.

Biofuels—such as ethanol from corn or sugarcane and biodiesel from oilseeds—have been increasingly heralded as possible substitutes. A renewable energy source, they can be produced domestically from a wide variety of plant materials and wastes. Because growing biofuels feedstock removes CO₂ from the atmosphere, biofuels should reduce GHG emissions relative to fossil fuels. They are sometimes also claimed to cause fewer emissions of various local air pollutants that are harmful to human health. The truth of these claims varies depending on the exact fuel blend, engine, and operating conditions. Biofuel emissions is a very complicated area with relatively few safe generalizations. Naturally the biofuels tend to have lower emissions of sulfur dioxide, but otherwise evidence is very mixed.

In addition to the environmental benefits, a secondary motivation for the promotion of biofuels is said to be rural economic development. Agricultural lobbies are strong and have been fairly successful in promoting for instance a combination of subsidies for ethanol production (supposedly as a climate policy), combined with policies to stop imports of Brazilian ethanol.

Since one of the biggest barriers to large-scale production of biofuels has been their higher economic costs compared to fossil fuels, government subsidies have long played an important role in the development of the biofuels industry. In the U.S., for example, the Energy Tax Act of 1978 introduced the first major federal subsidy to ethanol: a full exemption from the motor fuel excise tax. Since then, hundreds of government programs have been created to support virtually every stage of production and consumption—from growing the feedstock crops to the vehicles that consume the biofuels. So-called “renewable fuel standards” have been implemented to secure a minimum demand. They require that a specified percentage of biofuels is used. Finally, to protect domestic production specific tariffs on imported alcohol have been enacted (Rajagopal and Zilberman 2007).

As a consequence of the chain of subsidies and regulations, production capacity in the U.S. has grown steadily for more than two decades. Between 1980 and 1990, production levels of ethanol more than quintupled to 900 million gallons per year. This overall growth occurred despite some difficult times for the industry when oil prices were low, corn prices high, or both. Current industry expansion is frenzied, with annual production capacity jumping by nearly 40% between January 2005 and October 2006. In the case of biodiesel, the market remains much smaller than the market for ethanol (see Table 18-1), though its rate of growth has been faster.

The push for ethanol and other biofuels production by governmental policies has occurred not only in the United States (see Sorda et al. 2010). Brazil,

TABLE 18-1 World Ethanol and Biodiesel Production by Country (2008) in Million Liters.

<i>Country</i>	<i>Ethanol</i>	<i>Biodiesel</i>
U.S.	33,737	3,078
EU 27	2,748	9,164
Argentina	—	1,550
Australia	97	1,051
Brazil	24,261	1,238
China	1,882	—
Canada	892	114
Colombia	296	—
India	247	227
Indonesia	—	405
Malaysia	—	609
Thailand	337	—
Other	480	1,036
World	64,981	18,472

Source: Sorda et al. 2010.

which currently produces approximately the same amount of ethanol as the United States, derives almost all of it from sugarcane. Brazil—in fact the true pioneer—began its search for alternative energy very far back, even before the 1970s. Indeed, many studies about the use of neat vegetable oils, such as babassu, coconut, castor seed, and cotton seed in internal combustion engines go back all the way to the 1940s (see Pousa et al. 2007). In the 1980s, in response to 1970s petroleum shortages, Brazil's federal government created a program called PROALCOOL that implemented and regulated the use of ethanol as fuel. Through that program and many others, the government has offered economic incentives, set renewable fuel standards, invested in market promotion, and encouraged the auto industry to produce engines that can use biofuels. The Brazilian government is very engaged in the biofuel program, which seems to be an irreversible process. In 2007, the potential market for biodiesel was approximately 800 million liters, with expectations of reaching 2 billion liters by 2013.

In the case of the European Union, which mostly produces biodiesel from rapeseed, the substitution of biofuels for fossil fuels has been proposed as part of a strategy to mitigate greenhouse gas emissions from road transport, increase security of energy supply, and support development of rural communities. For that purpose, high (and perverse) tariffs on biofuels imported from more efficient producers (like Brazil) and high subsidies are in place. Currently, total support for biofuels in the EU is around €3.7 billion annually, of which €1.3 billion are devoted to ethanol (€0.74 per litre) and €2.4 billion are devoted to biodiesel (€0.50 per liter).

The Common Agricultural Policy (CAP) has also played an important role in the evolution of the biofuel industry within the European Union. In 1992, the MacSharry CAP reform introduced mandatory fixed shares of nonfood set-aside land. In 2005, more than 95% of the nonset-aside areas were allocated to energy crops (Sorda et al. 2010). Finally, many European cities have also played an important role promoting the use of biofuels by participating in research and demonstration projects and by using biofuels in their own fleet. For instance, as a voluntary initiative, the provider of public transportation services in Helsinki (HLK), in cooperation with the Helsinki Metropolitan Area Council, began to use a second generation biofuel in buses in 2007. Tax exemptions and financial support were also granted in the trial period (Silvestrini et al. 2010).

Although biofuels may help in the reduction of GHG emissions, whether they are desirable alternatives to fossil fuels depends largely on the type of energy crop and how they are produced. Most important, the use of biomass for fuel should be viewed within the framework of climate and energy policy as a whole. There is not enough biomass (nor land to grow sufficient biomass with current technology) to replace all fossil fuels. Technical analyses show that it is considerably more efficient to replace coal or oil in industrial, power, and heating applications with biomass than to substitute for transport fuels. The demands on the fuels for vehicle use are among the most exacting, and to take biomass and first convert it into a fuel that is good enough for a vehicle is a very expensive and roundabout manner to reduce fossil carbon emissions. Compare, for instance, using wood chips instead of coal in a cement factory or thermal plant (see, e.g., Azar et al. 2003 for evidence that the transport sector will likely be the last sector to make a transition away from fossil fuels, not the first).

Another important disadvantage of all bionergy sources is that they compete with food production. Conversion to biofuels makes up an increasingly rapidly growing demand for key feedstocks, raising concerns about the potential for food-based biofuels to be sustainable. Ethanol production burns up a significant fraction of U.S. corn supplies. Some accuse biofuel production for bringing inventories of corn to very low levels and pushing up the prices of basic crops. The Food and Agriculture Organization of the United Nations (FAO) reported that food prices skyrocketed by nearly 40% in 2007 and made further large jumps in early 2008. Nearly all agricultural commodities—including rice, maize, wheat, meat, dairy products, soybeans, palm oil, and cassava—were affected. Although a general increase in all raw materials and a weakened dollar were important factors, and bad weather in key production areas contributed to increased costs of production inputs and transportation, the role of biofuel policies in the food-price hikes has become particularly controversial since high food-prices could have severe consequences for poor countries and poor people and long-term irreversible consequences for health, productivity, and well-being.

It seems that the 2008 “fuel vs. food” debate was very exaggerated. There was little or no analysis of the supply side response, an otherwise prominent factor.

One might remember how Western countries are sometimes accused of ruining the preconditions for agriculture in developing countries by “dumping” agricultural surplus. The mechanism implied in this critique is that the dumped food lowers local food prices and reduces future incentives for farming. Naturally by this logic, it must be a great boost—at least in the intermediate run—to farming if richer countries start to import crops or crop-based products such as fuels. The concern about starvation due to biofuels might have been driven by very short-run concerns (before the supply side response). As a whole, it would seem to be an opportunity for agricultural countries if they find new profitable markets abroad—at least as long as they can expand production enough to both feed their population and still export some surplus. Those countries, regions, or individuals who are producers or net-exporters will gain, while those who are dependent on buying food will lose.

Biofuel production may also have negative environmental impacts. It can increase land degradation due to extensive use of fertilizers and pesticides; it can reduce water quality and create effluent runoff problems that affect downstream ecosystems; and it threatens biodiversity when natural lands are converted to productive lands. The impacts of biofuel production on land use changes may affect the capability of biofuels to reduce GHG. Biofuels from waste products can avoid land use change and its emissions. Algae or other biomass grown in the desert or feedstocks produced on lands that sequester little carbon today might also keep land use change emissions low, although the ability to produce biofuel feedstocks abundantly on unproductive lands is questionable. For the true long run, global calculation suggests that solar energy may be more promising. More than one order of magnitude more efficient than photosynthesis, solar energy can also be produced on nonfertile lands.

Moreover, growing crops and converting them to biofuels requires energy: to grow the feedstocks, power farm machines, produce farm machinery, fertilizers, and pesticides; transport the crops to biofuel production facilities; build and operate these facilities, and more.

Yet despite environmental damages and energy usage, recent work suggests some net benefit when biofuels replace petroleum products. However, the gains are moderate, especially for corn-based ethanol, and do not justify the level of the current subsidies. Some estimates indicate that ethanol subsidies in United States are well over US\$500 per metric ton of CO₂ equivalent removed. People argue that one could have achieved far greater reductions for the same amount of money by simply purchasing the reductions in the marketplace—e.g., 30 metric tons on the EU emissions trading system (ETS) (IISD 2006). Again we should remember that there may, in principle, be some important dynamic effects from new technology.

In sum, biofuel production has created important opportunities and challenges for sustainable development. It may play a highly positive role in mitigating climate change, enhancing environmental quality, and strengthening the global economy by creating opportunities for family farmers and rural

workers. However—with the current technologies—producing an important amount of energy with biofuels will require a large amount of land, potentially changing the landscape on Earth in a significant way. To avoid perverse outcomes, we need to pay attention to the effects of the development of the biofuel industry on the magnitude and direction of land use change elsewhere. The early identification of unintended consequences of alternative biofuels inputs may help to avoid costly mistakes and regrets about the environmental effects. This requires a systematic assessment of the full impacts of biofuel production on ecosystems and more coordination on the policies that are intended to promote its development.

Supplemental Reading

Murugesan et al. 2009
Schipper 2010

Thomas, V. M. 1995
Timilsina and Dulal 2011

Notes

- 1 Parts of this section are based on joint work with Åsa Löfgren and Henrik Hammar.
- 2 Leaded paints are banned in most countries. Lead batteries are subject to charges or deposit–refund schemes in some countries, including Sweden, where the military, in its environmental action plan, has decided to phase out the use of lead in bullets.
- 3 Actually, some lead was left for another few years, but the quantities were not significant.
- 4 This section builds on a number of World Bank reports and other papers (Lovei 1998; Onursal and Gautam 1997; Sayeg 1998; Shah and Nagpal 1997; Shah et al. 1997; UNDP/World Bank ESMAP 1998).
- 5 However, some countries (e.g., Panama and Mexico, at least during some years) let the oil companies charge more for unleaded gasoline because it costs more.
- 6 Anecdotal evidence seems hypothetical or even counterintuitive. Noticing that unleaded gas was more expensive, I asked a couple of Mexican taxi drivers if it was not a disadvantage to have a cab with a catalytic converter; they said they got more customers that way.
- 7 The World Bank’s Clean Air Initiative in Latin American Cities conference (Dec. 2–4, 1998) is one example (Lovei 1998; Onursal and Gautam 1997; Sayeg 1998; Shah and Nagpal 1997; UNDP/World Bank ESMAP 1998).
- 8 This situation means that there are two parallel avenues for improvement. Not only have the market shares for Classes 1 and 2 been increasing, but the requirements for each have become increasingly stringent over time. Currently, lead content is required to be <5 milligrams/liter of fuel; sulfur <50 ppm and <100 ppm for Classes 1 and 2, respectively; and benzene <1%. Some technical standards (e.g., vapor pressure) also vary geographically and seasonally.

Vehicle Standards, Urban Planning, and Lessons Learned

Aside from fuel quality, the main environmental policy options available to deal with the environmental problems caused by transportation are vehicle standards (for new vehicles but also the inspection and maintenance of older vehicles), car scrapping, traffic management, logistics, and urban planning. In this chapter, we describe such policies and the lessons we have learnt so far.

Vehicle Standards, Efficiency, and Distributional Concerns

When it comes to vehicle control, most worldwide attention is paid to new vehicles. New vehicle control or standards date back to the late 1960s, and in the United States, federal emissions standards for new vehicles requiring catalytic converters (among other things) went into effect in the mid-1970s. Indeed, the U.S. Clean Air Act and its various amendments have been a prominent driving force behind technological progress, but there have also been important initiatives from Japan and some European countries. In 1990, California took the lead when the California Air Resources Board (CARB) adopted a visionary scheme of progressively tighter emissions requirements, and vehicle manufacturers were forced to sell minimum shares of vehicles fulfilling successively tighter environmental criteria (see Table 19-1). For instance, 25% of vehicles manufactured after 1997 had to be low-emission vehicles (LEVs).

Several details of the CARB programs have been modified, leading to the development of the so called “LEV II” Program and the “LEV III” Program, but it has undoubtedly been an important mechanism for encouraging new technological developments (California EPA). In 2004, the State of California also introduced, as part of the LEV II Program, GHG emission requirements, which have proven to be precursors of the proposed U.S.-wide joint regulation of U.S. EPA and Department of Transportation on mutually compatible GHG emission standards and fuel economy standards (see Chapter 17). The current legislation aims for 10% zero-emission vehicles (ZEVs) for the period

TABLE 19-1 Overview of CARB Regulations in 2010, for Passenger Cars and Light-duty Vehicles, Standard Gasoline

<i>Program</i>	<i>Vehicle emission category</i>	<i>Requirements (grams/mile)</i>	<i>Minimum percentage of vehicles</i>
LEV II	LEV (2004–2010)	HC<0.018 NMOG<0.09	25% for 2004
		NO _x <0.07 CO<4.2	50% for 2005
	ULEV	HC<0.011 NMOG<0.055	75% for 2006
		CO<2.1 NO _x <0.07	100% for 2007–2010
	SULEV	HC<0.004 NMOG<0.01	
		NO _x <0.02 CO<1.0	
ZEV	ZEV	0 (locally)	10% for 2005–2010
			11% for 2009–2011
			12% for 2012–2014
			14% for 2015–2017
			16% for 2018 onward

Notes: LEV = low-emission vehicle; ULEV = ultra-low-emission vehicle; SULEV=super-ultra-low-emission vehicle; ZEV = zero-emission vehicle; HC = hydrocarbon (formaldehyde); NMOG = nonmethane organic gases; NO_x = nitrogen oxides.

2005–2010, after which the target will increase gradually to 16% in 2018. For the near term, part of this obligation can be fulfilled by credits earned from partial ZEVs (PZEVs). Early introduction of ZEVs (before 2003) is an added “flexibility mechanism” that gives additional credits. Whereas standards such as the LEV are regulations, they are not as inflexible as a design standard (such as mandatory catalytic converters) because they allow manufacturers to choose how to meet the standard (see Chapter 4). Despite dramatic improvements in fuel, engine design, and emissions reduction, the most promising future in this area probably belongs to other technologies, such as electric and hybrid vehicles¹. The U.S. government and others have tried to promote alternative fuel vehicles (AFV) for a long time. The 1992 Energy Policy Act established the goal of having alternative fuels replace at least 10% of petroleum fuels in 2000 and at least 30% in 2010 (Romm 2006). Nevertheless, historically there have been major barriers to AFV success—among them, the high initial cost of the vehicle, safety and liability concerns, and high fueling cost (compared with gasoline).

Many developing and recently industrialized economies—such as Brazil, Hong Kong, Malaysia, Mexico, Republic of Korea, Singapore, Taiwan, and Thailand—have adopted regulations for new vehicles (and for fuels) that are not far behind those adopted in the United States and Europe. The vehicle industry exerts considerable pressure because it prefers standardized requirements in all markets and because standardization gives large international vehicle manufacturers a competitive edge over small local producers. This pressure may give rise to tensions vis-à-vis local national interests. Although these developments

are encouraging, from an environmental standpoint at least, they do not have as much effect as in industrialized countries for several reasons: in developing countries, vehicles have a long lifetime and are poorly maintained, many used vehicles are imported, and a large fraction of the vehicle pool are two-stroke engines (e.g., motorcycles, mopeds, and three-wheeled tuk-tuks).

The two-stroke engines warrant special attention. Even though technological solutions such as computer-controlled fuel injection (in fact, many of the technologies used in four-stroke engines) could be applied to two-stroke engines, some problems related to lubrication remain because the lubricating oil is a source of dangerous emissions. Also, two-stroke engines typically do not have sophisticated technology; they are cheap and operate under rough conditions with little maintenance. Many other kinds of equipment with two-stroke engines also deserve attention, such as chain saws, machinery for road building and agriculture, and motorboats. Some countries have already banned certain high-polluting vehicle types. For instance, the government of Bangladesh promulgated an order banning all two-stroke engines from the roads in Dhaka starting on December 31, 2002. Evidence suggests that fine particulate matter and black carbon concentrations decreased significantly due to the ban (Chowdhury 2005).

Another solution is to require the use of ultraclean fuel (e.g., alkylate petrol, which consists essentially of straight alkanes) for two-stroke engines, which have inherently poorer combustion properties². Again, the complex interplay between fuel and motor regulation is highlighted. Sometimes cleaner fuels (with alkanes or alcohol) can serve as a substitute for cleaner combustion; sometimes they are complements (as with lead elimination, necessary for the operation of catalytic converters).

Inspection and Maintenance for Used Vehicles

Despite technological and regulatory advances, new vehicle standards are not sufficient to achieve ambient goals (if vehicles deteriorate rapidly or owners keep old cars instead of scrapping them). In fact, draconian standards on new cars combined with lax regulation on older vehicles would be a good example of new source bias (see Chapters 9 and 13) and could have a negative effect on the ambient environment if it led to a significant delay in the turnover rate of the vehicle stock. In Japan, inspection and maintenance requirements become extremely stringent when cars are more than three years old, so the vehicle stock turns over rapidly. The used cars are exported en masse to neighboring countries. This approach benefits at least some interests in Japan: The auto industry gets high sales figures, and the local environment is subjected to fewer and fewer emissions.

U.S. EPA has encouraged states to establish inspection and maintenance programs to maintain a minimum standard for vehicles on the road. Because a

large share of pollution comes from a few vehicles, concentrating on the most polluting vehicles should improve ambient air quality efficiently. Although the potential gains of inspection and maintenance programs are large, the practical results to date are not impressive (Harrington and McConnell 1999). In 1990, the U.S. Congress established in the Clean Air Act more stringent requirements for enhanced inspection and maintenance programs, which started to be implemented in 1995. Particularly in the United States, overseeing such a program involves a principal–agent structure with several stages. U.S. EPA monitors the state programs and requires them to include various features in their enhanced inspection and maintenance programs. Compliance gives the states “credits” toward the fulfillment of their state implementation plans, which help determine whether the state (or parts of it) has designated “nonattainment areas.” The states act as agents vis-à-vis the federal agency (U.S. EPA) and as principal vis-à-vis the local controlling agents down to the individual motorists.

The U.S. programs fail to deliver emissions reductions at reasonable costs for several reasons (Harrington and McConnell 1999):

- The assignment of liabilities to the individual motorist results in high transaction costs.
- Most of the resources in the program are used for emissions testing and little for actual repair. Failure rates at tests are low, suggesting that more selective testing might be beneficial.
- Motorists have many opportunities to evade required repairs. They can test vehicles numerous times, until they happen to “get lucky” and pass. Many jurisdictions (outside large cities) do not require emissions testing as part of inspection and maintenance; hence, vehicles can be sold to someone or even easily (although illegally) registered in such an area. In Arizona, 22% of vehicles that fail the emissions test never come back for a retest (Ando et al. 2000). It is not clear where they go, but presumably, some go to areas that do not require a test, or the owners drive the vehicles illegally.
- Allegedly, some mechanics specialize in temporarily “fixing” vehicles to pass the emissions test with little or no permanent effect.
- The system includes numerous “waivers.” A motorist who is over a certain age and drives few miles may be totally exempt, and a motorist who can prove having spent a certain amount of money (not much—maybe \$50, \$75, or even \$15) on repairs may not be required to make further repairs, even if the car fails the emissions retest.
- Waivers for cars driven short distances make sense because the damages are small. Income distribution is also important, because the oldest and dirtiest cars tend to be cheapest and are owned by the poor. More stringent requirements may hit vulnerable groups. However, if these vehicles have high mileage (and are driven mostly in inner cities), then the environmental damage may be considerable. In such cases, it does not make sense to help low-income people by absolving them from these important environmental standards.

- Limitations in enforcement reflect negatively on program efficiency, and as a result, dirty cars are seldom found and only rarely repaired properly (or retired). In the worst cases, the inspection and maintenance program degenerates into being a hassle for owners of new and fairly clean cars, for which the program provides little environmental benefit. U.S. experience shows that vehicle testing should generally be separated from repair. Combined “test and repair” facilities have incentives that are far from efficient. Also, mandating enhanced inspection and maintenance with a tougher mandatory minimum repair cost of more than \$450 was difficult and led to considerable public resistance, ultimately causing U.S. EPA to back down. On the other hand, other mechanisms in society may compensate for the lack of formal monitoring. In California, using citizens’ complaints in the search for the worst-offending vehicles appears to have been successful (see Box 19-1).

Most other countries have inspection and maintenance programs. In the Swedish program, all cars more than a couple of years old are inspected every year in an integrated environmental and safety inspection and maintenance program³. Sweden is not a society in which unregistered cars can be driven for very long; monitoring is rather strict, and cars with deficient safety or

BOX 19-1 CUT-SMOG Smoking Vehicle Program

California’s South Coast Air Quality Management District (AQMD), the smog-control agency for all or portions of Los Angeles, Orange, Riverside, and San Bernardino counties, has regulated regional air emissions since 1987. The AQMD developed the Smoking Vehicle Program to reduce visible exhaust from vehicles in the area and established a complaint “hotline” to which people could voluntarily and anonymously report (by e-mail or by calling AQMD’s toll-free number) vehicles emitting excessive amounts of exhaust.

The AQMD cannot impose fines, but it does send a letter to the owner of each reported vehicle. The letter advises the owner that the vehicle was reported and recommends having it repaired. The letter also advises the vehicle owner that excessively smoking vehicles can be fined.

By 2000, nearly 1 million people had volunteered information to the CUT-SMOG program. On average, more than 40% of the “offending” vehicle owners have returned compliance forms to the AQMD, claiming that they have had their vehicle inspected and repaired, when necessary.

In 2010, AQMD launched the nation’s first fully featured smart-phone application for a wide range of air quality information. The free *Air Quality* app, designed for the iPhone, iPad, and iPod touch, features real-time and forecasted air quality levels for user-selected areas as well as interactive maps of alternative fuel filling stations, air quality news, and other features to help promote healthy living.

Source: AQMD 1996, 2010, <http://www.everythinglongbeach.com/check-air-quality-cut-smog-with-aqmd-iphone-app/>

emissions tests will definitely be stopped. There are no cost-based or other “waivers” (except for antique cars or museum pieces). The Swedish program is thus likely to be effective in stopping dirty cars, although at a high cost—expenses of inspection, waiting, maintenance, repairs, lost years of use from scrapped vehicles, and so forth—particularly for cars that get driven short distances in the country, where the potential damage to the environment is minimal. Concerns relating to income distribution may be less applicable in Sweden because income is not so unevenly distributed and public transportation readily available; the (relatively) poorer vehicle owners are not so dependent on old, polluting personal vehicles, and they can take public transportation if needed. The share of income spent on vehicles and fuel does not vary strongly with income in Sweden.

Although U.S. and Swedish experiences differ, they have some traits in common. These programs have significant transaction costs due to the fact that liabilities are assigned to the individual motorist, who has an incentive to avoid the maintenance needed for better environmental results. Other possible assignments of liability or other instruments could help alleviate the principal-agent aspect of this monitoring problem (Harrington and McConnell 1999):

- The liability could be forced back onto the vehicle manufacturers to a greater extent. This happens in other industries, such as the airlines, which lease engines from manufacturers, who are thus forced into taking maintenance into account. Such a tendency is observed for vehicles, too—with demands for extended emissions warranties covering more years and miles. This has led to much lower emissions at 50,000 miles for 1991 vehicles than for those made earlier. However, problems of moral hazard enter into extending an unlimited warranty: because the manufacturer has no control over fuel quality or everyday maintenance, the owner of a really old vehicle may have an incentive to damage the vehicle if someone else is liable for all repairs. Furthermore, long liabilities may produce practical difficulties (e.g., car manufacturers disappear from the market).
- Liability could be centralized, either through leasing (rather than purchasing) vehicles or through the separation of vehicle ownership from emissions liability, whereby the latter could be auctioned off (for whole classes of automobiles) to specialized companies that would receive payment from the government in exchange for taking care of, say, “all 1992 Fords.” Performance would be monitored by remote sensing, which is not precise for individual cars but sufficiently exact for large numbers of randomly sampled cars. These specialized companies would have to determine how to entice individual owners to inspect and repair, and they would have a clear incentive to do so efficiently.
- With individual liability, the monitoring problem could be reduced by shifting the burden of proof through a deposit-refund type of mechanism, such as a high annual fee that could be reduced (refunded) on

proof of low emissions values (possibly also on other grounds for exemption, such as the use of vehicles only on a farm or in the country). This approach changes the monitoring and repair incentives. Singapore uses a combination of high registration and annual taxes with rebates for certain categories (such as weekend cars) or for owners who replace old cars with new ones (see Chapter 16).

- The last of these alternatives might have the greatest relevance in the polluted megacities of developing countries, where the car stocks are commonly old. They urgently need efficiency but cannot afford inspections that do not lead to improvement, so better targeting is needed. Income distribution effects vary between countries, but in the poorer countries, private cars are luxury goods; hence, concerns should be limited. Still, among motorists, the poorer will almost inevitably have the most-emitting vehicles. This does not necessarily mean that those motorists lose out from a long-run policy of requiring maintenance; the rising costs of maintenance will presumably tend to be “capitalized” in a lower purchase price. This effect means that the former owners of the vehicle—including the first owner—share in the incidence of these expected costs. However, when new rules are made in periods of transition, those who have bought vehicles that they later find out are in need of emissions-related repairs—or, in the worst-case scenario, cannot be driven at all—definitely have a lot to lose.

Car Scrapping

Car scrapping programs are intended to simultaneously serve two goals. First, governments aim at reducing emissions and fuel consumption by providing a subsidy to those consumers replacing older, polluting vehicles with newer, cleaner, fuel-efficient ones. Second, since these programs encourage purchases of new automobiles, they stimulate the development of the automobile industry and employment in that sector. Indeed, due to the negative effects of the 2008 economic crisis over the automobile industry (sales of new automobiles decreased by 35% in the United States and 9.3% in Europe) and growing concerns about CO₂ emissions, car scrapping programs have been implemented in several European countries and the United States during recent years (see Table 19-2). (The strong incentives for “environmentally friendly” cars in Sweden discussed in chapter 16 are also relevant in this context).

For instance, the car scrapping scheme in Germany, the most successful program in Europe, had the declared main purpose of supporting the automotive industry and employment in the sector while achieving environmental targets, such as reduction in air pollution. The environmental subsidy of €2,500 per car was paid from funds allocated by the Federal Government of Germany (as part of the Economic Recovery Program II) to car owners that

TABLE 19-2 Overview of Car Scrapping Schemes in Europe and United States, 2009–2010.

Country	Period	Incentive	Vehicle Age	Conditions
Austria	4/1/2009	€1,500	>13 years	• Minimum Euro 4 • Dealers pay 50% of the incentive • Total budget €45 million (exhausted)
	12/31/2009			
Cyprus	2009	€675–1,700	> 15 years	• €1,280 if new car consumes less than 7 liters/100 km; €1,700 if new car consumes less than 5 liters/100 km.
France	12/4/2008	€1,000	>10 years	• New car emits maximum 160 g/km of CO₂ • No CO₂ emissions requirement for LCV • Total budget €380 million
	12/31/2009			
	1/1/2010	€700	>10 years	• Passenger cars + LCV⁽¹⁾ • New car emits less than 160g/km of CO₂ • No CO₂ emissions requirement for LCV
	6/30/2010			
Germany	7/1/2010	€500		
	12/31/2010			
Germany	1/14/2009	€2,500	>9 years	• Minimum Euro 4 • Maximum 1 year old • Total budget €5 billion (exhausted)
	12/31/2009			
Greece	9/28/2009	€500–2,200	>5 years	• Subsidy if individuals scrapped a car or LCV and did not purchase a new one.
	11/2/2009			
		€1,500–3,200		• Subsidy if individuals scrapped a passenger car and purchased a new Euro 4 or 5 car.
		€2,000–3,700		• Subsidy if individuals scrapped an LCV and purchased a new Euro 4 or 5 one.
		€3,000–9,000		• Subsidy if individuals scrapped a truck and did not purchase a new one.
		€3,000–7,000		• Subsidy if individuals scrapped a truck and purchased a new one newer than 5 years.

TABLE 19-2 (Cont.)

Country	Period	Incentive	Vehicle Age	Conditions
Ireland	1/1/2010 12/31/2010	€1,500	>10 years	• New car emits maximum 140g/km of CO₂
Italy	2/7/2009 12/31/2009	€1,500–5,000 (cars) €2,500–6,500 (LCVs)	>9 years	• Minimum Euro 4 + emits maximum 130 g/km (diesel) or 140 g/km (other fuels) of CO₂ • Subsidy can be combined with additional subsidies (€1,500 for new car running on CNG, electricity or hydrogen; increased to €3,000 if it emits exactly 120 g/km and to €3,500 if it emits less than 120 g/km; increased to €2,000 if the car runs on LPG and emits less than 120 g/km). • Subsidy for LCVs can be combined with purchase incentive of €4,000 if running on CNG. If the new LCV runs on LPG, electricity or hydrogen, the subsidy is increased to €2,000 if the vehicle emits less than 120 g/km.
Luxembourg	1/22/2009 7/31/2010	€1,500–2,500	> 10 years	• €2,500 if new car emits maximum 120 g/km of CO₂ (diesel ≤ 5 mg PM) • €1,500 if new car emits maximum 150 g/km of CO₂
The Netherlands	2009–2010	€750–1,000 €1,000–1,750	> 13 years > 9 years	• Petrol cars/LCV €750 if older than 19 years; €1,000 if older than 13 years (and newer than 19 years). • Diesel cars/LCV €1,000 for old car; €1,000 for old van < 1,800 kg; €1,750 for old van > 1,800 kg • New car/van equipped with particulate filter

Funds exhausted in March 2010.

continued

TABLE 19-2 (Cont.)

Country	Period	Incentive	Vehicle Age	Conditions
Portugal	1/1/2009	€1,000	> 10 years	• New car emits maximum 140 g/km of CO ₂
	8/7/2009	€1,250	> 15 years	
	8/8/2009	€1,250	> 8 years	• New car emits maximum 140 g/km of CO ₂
	12/31/2009	€1,500	> 13 years	
	4/1/2010	€1,000	> 10 years	• New car emits maximum 150 g/km of CO ₂
	NA	€1,250	> 15 years	
Romania	2/1/2009	Approx €900	> 10 years	• Maximum 60,000 cars, only 65% of funds available were used
	12/31/2009			
	2/19/2010	Approx €950	> 10 years	• Purchases of a maximum of 3 cars (max €2,850) • Total budget €57 million • Initial limit 60,000 cars (40,000 for private persons, 20,000 for companies), supplemented in March 2010 with 50,000 cars
Spain	12/1/2008	Interest-free	> 10 years or >	• Maximum price €30,000
	10/1/2010	loan up to €10,000	250,000 km	• New car emits maximum 140 g/km of CO ₂ • New LCV emits maximum 160 g/km of CO ₂ • Also purchases of used cars (up to 5 years old)
	5/18/2009	€2,000	> 10 years	• Manufacturers pay 50% of the incentive
	5/18/2010		(purchase new)	• New car emits less than 149 g/km • New commercial vehicle emits less than 160 g/km
			> 12 years	• New vehicle value price €30,000 • Applicable to used cars up to 5 years old • Maximum 200,000 vehicles

TABLE 19-2 (Cont.)

Country	Period	Incentive	Vehicle Age	Conditions
Slovakia	3/9/2009	€1,000–1,500	> 10 years	- Maximum price €25,000 €1,000 incentive without contribution from the dealer. €1,500 incentive when dealer contributes €500 (total customer benefit in that case €2,000).
	3/25/2009			
United Kingdom	4/6/2009	€2,000	> 10 years	- Dealers pay 50% of the incentive - New car value maximum €25,000
	12/31/2009			
United Kingdom	5/1/2009	£2,000	10 years	- Manufacturers pay 50% of the incentive - Passenger cars and light commercial vehicles - Total budget £400 million
	3/31/2010			
United States	7/1/2009	US\$3500–\$4500	Fuel Efficiency	- Subsidy is based on fuel efficiency. - US\$3500 if new car increased fuel economy ranged between 4 and 10 miles per gallon. - US\$4500 if new car increased fuel economy was larger than 10 miles per gallon. - Total budget US\$3 billion. - Funds exhausted in August 2009.
	11/1/2009			

Note: 'LCV = Light commercial vehicles.

Sources: European Automobile Manufacturers' Association, 2010 and National Highway Traffic Safety Administration, U.S. Department of Transportation, 2009.

scrapped a car older than nine years and purchased a car newer than one year fulfilling at least the Euro IV emission standards. The program, launched in January 2009, had an initial budget of €1.5 billion, and the initial limit of 600,000 cars was reached in merely three months. The program was extended, and the final total funding amounted to €5 billion, covering approximately 5% of all registered cars in Germany.

Initial estimates indicate that the newly produced cars acquired within the Car Scrapping Scheme release significantly less emissions than the older scrapped cars; in aggregate, the replacement of 2 million old cars in 2009 led to a reduction of 9% of benzene emissions, 7% reduction of CO₂ and NO_x emissions, 5% reduction in particulate matter, and an overall 4% decrease in total emissions by passenger cars (FMENCNS 2009). Nevertheless, estimates also indicate that these reductions could have been even larger, since the lack of emissions standards in the awarding of the scrapping subsidy led to purchase of cars with rather high emissions levels relative to the lowest emissions available alternatives.

In the United States, the Consumer Assistance to Recycle and Save Program (CARS) granted owners of vehicles meeting statutorily specified criteria a monetary credit for trading in their vehicle and purchasing (or leasing) vehicles with increased fuel efficiency. New vehicles with an increased fuel economy ranging between 4 and 10 mpg earned a credit of \$3,500; if the new vehicle's increased fuel economy exceeded 10 mpg, the credit was \$4,500.

Unofficially dubbed "Cash for Clunkers," the program was initially allocated \$1 billion and was scheduled to run between July and November 2009. Nevertheless, due to the unexpectedly high demand, the initial funds were depleted by the end of July 2009. An additional \$2 billion allocated to the program was depleted again by August 2009. In total the program resulted in over 670,000 cars being traded in. Because of the unanticipated strength of consumer response, the program led to a sharp decline in dealer inventories and caused several major manufacturers to increase production schedules through the end of 2009, leading to an increase in employment and GDP. The estimated immediate gross impact on GDP of all transactions was \$7.8 billion from the sales of the entire CARS fleet (U.S. DOT 2009).

As intended, the program led to reduced fuel consumption, resulting in a corresponding decrease in fuel tax revenues to both federal and state governments.

Estimates indicate that the mean fuel consumption for the new vehicles was 34% lower than the cars traded in. This decrease led to a reduction in fuel tax revenue of about \$345 million.

Based on the estimated reduction in fuel consumption, total reduction in CO₂ emissions as a result of the CARS program was roughly 9 million metric tons, which represents 0.15 percent of annual U.S. carbon emissions. In addition, other pollutants—particulate matter, nitrous oxides and sulfur dioxide—were also reduced.

Both the U.S. and German programs succeeded in encouraging a reduction of air pollutants in the short run. However, their effects have to be regarded carefully, since most of the purchase decisions would have occurred in the nearest future due to the normal life-cycle of cars, regardless of the subsidies. Thus, the real contribution of these programs is to speed up the switching process, inducing an earlier transition to “environmentally friendly” automobiles. On the other hand, their global effect in reducing CO₂ emissions is ambiguous since they have negative effects on the secondhand commercialization of cars in developing countries. For instance, many of the cars scrapped in Germany or United States would have been exported to developing countries, where they might have the opposite effect.

Low Emission Zones

In 2005, the European Union enacted the 2005 Clean Air Directive by which emission standards were set for urban areas of European cities, in an attempt to limit the harmful effects of traffic pollution on human health. If the maximum allowable limits are exceeded, local authorities must develop clean air action plans which include infrastructure developments such as ring roads, public transportation, or the implementation of Low Emission Zones (LEZ) where highly polluting vehicles are banned or in some cases charged if their emissions are over a set level.

As of April 2010, Europe already has 75 LEZ schemes in place, with 19 schemes confirmed or scheduled to start in coming months or years. They have been implemented in Austria (1 operating, 1 planned), Czech Republic (1 operating), Denmark (3 operating, 2 planned), Germany (43 operating, 2 planned), Hungary (1 planned), Italy (8 operating), the Netherlands (12 operating, 9 planned), Norway (3 planned), Sweden (5 operating), and the United Kingdom (2 operating, 1 planned).

Most LEZ schemes in place have the following features:

- They operate all year around.
- They mainly ban heavy diesel-powered trucks, buses, and coaches that do not fulfill Euro standards. Nevertheless, programs differ within European countries with respect to the level of stringency of the Euro standard that is required and the type of car that is included. For example, in the Netherlands, all heavy-duty diesel-engined pre-Euro 3 vehicles are banned from circulating in the city centers, while the German schemes cover both trucks and passenger cars pre-Euro 1.
- The cars that are not banned must have a sticker displayed on the windshield. Foreign vehicles are also enforced and are required to have a sticker.
- Usually, the schemes are enforced manually by the local police, although some countries already have electronic systems reading vehicle number plates.

Estimates indicate that the implementation of LEZ has been very successful in reducing emissions, although the total effect is clearly dependent on the heaviness of the traffic, the stringency of the standard, and the amount of dirtier vehicles travelling in the area. For instance, the Stockholm LEZ reduced the emissions of particulate matter (PM₁₀) by about 60% and the emissions of nitrogen oxides by 20% during the period 1996–2000. In the case of Berlin, where a much less stringent Euro standard is required to enter the urban areas, the LEZ reduced emissions of diesel particulates by 24% and PM emissions by only 8% (LEZEN 2009).

Urban Pollution in Developing-world Cities

In many of the large cities of developing countries increased traffic-generated air pollution is drawing increasing concerns (Han and Naeher 2006). In China, for example, passenger cars and large vehicles (including heavy duty trucks and buses) are the major sources of urban air pollutants like NO_x and particulate matter (Wang et al. 2010).

Poor countries always have a heightened need for prioritization: The number of severe problems is great, the resources to tackle them small. Much can be said for integrated urban pollution management programs in such cities as Mexico City, Mexico; Santiago, Chile; Dhaka, Bangladesh; Cairo, Egypt; and São Paulo, Brazil. The experiences, particularly from Latin America, have shown that four broad strategies need to be pursued (Faiz et al. 1995; Takeuchi et al. 2007; and Narain and Krupnick 2007):

- Improve vehicle technology through engine replacement, inspection and maintenance, and vehicle scrappage;
- Use cleaner (reformulated or alternative) fuels that can be targeted (by price policies) at particular areas of application;
- Control and manage traffic; and
- Improve urban transportation infrastructure.

In practice, the number of steps that can be taken to improve the air in a city like Mexico City is large (Eskeland 1994; Goddard 1997). By organizing these steps in ascending order of marginal cost, an engineering estimate of the marginal cost of abatement curve can be produced for the whole city (see Figure 19-1). At first, improvements include retrofitting trucks and minibuses with liquid petroleum gas and recovering gasoline vapors at gas stations and depots at low or even negative cost⁴. Then come inspection and maintenance for various kinds of vehicles—starting with high-use vehicles such as taxis or delivery vans, moving gradually to private passenger vehicles. Other measures include fuel improvements (tighter standards, removal of impurities, the inclusion of certain additives, and lowered vapor pressure), paving of roads, new

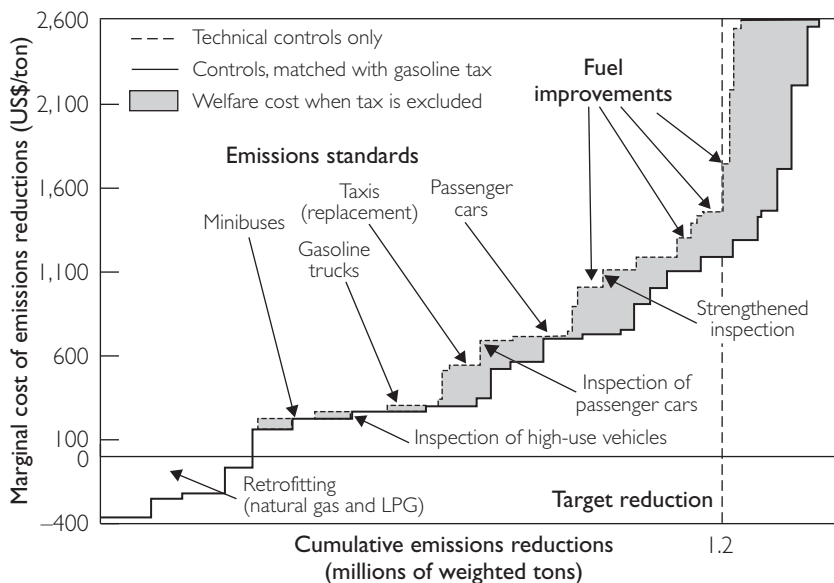


FIGURE 19-1 Engineering Estimate of Abatement Costs in Mexico City, 1992 Estimates

Source: Eskeland 1994.

vehicle standards, and so forth. Some improvements can be done to various degrees with increasing marginal costs (e.g., inspection and maintenance has a high cost-efficiency for professional vehicles, but not when extended to all vehicles). The most difficult but not necessarily least important are the most systemic improvements, such as transportation control and zoning measures.

Considerable savings are possible through a good combination of policy instruments, including a fuel tax and a combination of physical reduction measures; without the gasoline tax, an otherwise well-designed program of controls would be almost 25% more expensive (or US\$110 million/year in Mexico City) (Eskeland and Devarajan 1996). This is a good example of a case in which two policy instruments may be combined in a “second-best” strategy (given that environmentally differentiated road pricing is unavailable). The fuel tax helps reduce overall demand, whereas the technical improvements reduce emissions per mile. In addition, the tax proceeds can be used to finance abatement. Generally, economists emphasize the inefficiency of such earmarked funds. In some institutional settings (where the environmental authorities would otherwise not be able to compete for any funds), this revenue recycling might provide additional arguments for pollution fees.

Somewhat similar results were found for Manila, Philippines, a city of 10 million where 80% of the population lives in areas where the national standards for total suspended particles (TSPs) are exceeded (Shah and Nagpal 1997). TSP levels are frequently more than five times the WHO guidelines. Some measures

TABLE 19-3 Benefits and Costs for Several Policies in Manila, Philippines

<i>Abatement measure</i>	<i>Avoided effects</i>	<i>Benefits (million US\$)</i>	<i>Costs (million US\$)</i>	<i>Approximate benefit/cost ratio</i>
Address gross polluters	160 deaths, 4 million RSD	16–20	0.08	20–25
Clean vehicle standards	895 deaths, 24 million RSD	94–116	10–20	7
Vehicle inspection and maintenance	310 deaths, 8 million RSD	30–40	5.5	6
Improved diesel quality	94 deaths, 2.5 million RSD	10–12	10	1

Note: RSD = respiratory symptom days.

Source: Adapted from Shah and Nagpal 1997 (table ES2).

can be 20–25 times as cost-effective as others (Table 19-3). One key measure is overall traffic planning and management. Because calculations are inherently difficult, we simply present one good example and one bad example.

The biggest future challenges are likely to come from the desire for increased mobility from the bulging middle classes of countries like China and India. Both these countries are building roads and car factories at very high rates. The development of ultra-cheap cars such as the Tata Nano may well herald a new era. Even though it has won many awards for good marketing and good design, and even though it appears (after some further effort) to be able to meet high environmental standards, the Tata Nano will still represent a major challenge. Its fuel efficiency is good (around 25 km per liter)—roughly twice that of other small cars. However, at a price of little more than US\$2000, it could invoke a surge in car sales that would lead to considerable increases in congestion, total fuel use, and carbon emissions.

Traffic Management through Rationing that Backfires

One policy widely considered an abysmal failure is the “day without a car” (“hoy no circula” or “día sin carro”) policy implemented in Mexico and in many major cities in other countries. It was designed as a rationing scheme by which each car was banned from driving one weekday. Several appealing features of this plan were touted:

- Compliance is easy to monitor; the license plate number indicates which day the vehicle is banned. (Anecdotal evidence suggests that some people might have had two plates, but monitoring was simple—partly because the way the police monitor can in turn be monitored and complemented by any casual observer.)
- The program is “fair” because it targets the rich and poor equally.
- Commuters who travel only by car supposedly will learn about the public transportation system available to them and perhaps realize its benefits.

In fact, none of these expected features turned out as planned. Instead, problems inherent in the program became apparent (Eskeland and Feyzioglu 1997a, 1997b; Davis, 2008).

Davis (2008) compared air quality before and after the restrictions were implemented using high-frequency measures of five major air pollutants from monitoring stations. Across pollutants and specifications, no evidence showed that the program improved air quality. Instead, since drivers put off trips to weekends and nonpeak weekdays, the policy engendered a relative increase in air pollution during those days. No evidence of an absolute improvement in air quality during any period for any pollutant was apparent.

Whereas it was hoped that the driving restrictions would cause drivers to substitute to low-emission means of transportation, there was also no evidence of increased ridership of the subway, public bus system, or private bus system. Instead, evidence from vehicle registrations and automobile sales indicates that the program led to an increase in the total number of vehicles in circulation as well as a change in the composition of vehicles toward used, and thus higher-emitting, vehicles. From 1983 to 1989—before the regulation—Mexico City exported used cars to the rest of the country at a yearly rate of 74,000 used vehicles per year (typically replaced by newer ones because average income is higher in the capital). Immediately after the introduction of the program, however, the capital imported 85,000 used vehicles from the rest of the country.

The regulation should not necessarily be seen just as a ban on one day a week but rather as the issue of (nontradable) driving permits for six days a week. The allocation mechanism of six permits per vehicle created a perverse incentive to acquire additional vehicles (an example of a perverse output effect; see Chapter 11). Overall, fuel demand, congestion, and pollution were all increased by the policy (Eskeland and Feyzioglu 1997a, 1997b; Davis 2008).

Moreover, there was nothing particularly “fair” about the policy. In fact, the wealthier households appeared to be the ones who had the greatest ease in getting around the regulations by buying extra cars. Overall, the pattern of behavioral responses indicates that the restrictions were unsuccessful in inducing drivers to substitute away from private vehicles.

Traffic Management through Foresight and Planning

One example of rare foresight in town planning is in Curitiba, the capital of Paraná, Brazil, where local authorities have long insisted on sustainable transportation and have consistently favored buses over heavier, more expensive rail (Meadows 1995; Prefeitura Municipal de Curitiba 2002; Rabinovitch 1992; Tlaiye and Biller 1994). While not a megacity, Curitiba is large and growing quickly: population went from 0.5 million in 1965 to 1.75 million in 2008. What sets Curitiba apart is its long-term commitment to make Curitiba “livable” with ample green space, reasonable garbage handling, and sensible traffic management.

The original plan came out of a public competition in 1965. Authorities then set up the Curitiba Research and Urban Planning Institute. Coherent town planning and zoning limits developments in the center, which has large pedestrian zones and emphasizes cultural heritage. The first express busways were inaugurated in 1974. Instead of the usual mix of unstructured roads and expensive rail found in most world cities, Curitiba has followed an almost Napoleonic vision of planned road construction. The hierarchy of roads is based on the central linear axes with a three-way structure: at the center are exclusive express bus lanes, flanked by two local roads. On each side of the central road (a block away) are high-capacity one-way roads for traffic into and out of the city. These major arteries are complemented by a system of four levels of roads (labeled “structural,” “priority,” “collector,” and “connector”), the smallest of which are the minor residential roads.

One of the factors tipping the balance in favor of express buses rather than rail systems is simply the capital costs. Curitiba’s direct-route busways have a construction cost of less than \$0.2 million/km compared to \$20 million to \$100 million/km for underground or aboveground rail systems. To be able to choose this alternative, city planners have to acquire the necessary land before the city gets large and congested—a necessary aspect of long-term planning.

Currently, Curitiba’s public transportation system carries more than two thirds of the population each day. It is operated by private bus companies that are selected by competitive bidding and work on tightly regulated concessions with the municipal government. The bus fares go to a municipal bus fund, and the companies are paid per kilometer served. Even though Curitiba has a high rate of car ownership (more than most major cities in Brazil), there is no serious congestion, and public transportation is estimated to save 25% of fuel consumption. Several other large cities are trying to copy the Curitiba concept, and some find it difficult precisely because they feel the infrastructure should have been put in place before the cities grew to their current size. However, this is no cause for delay. Most forecasts show continued dramatic urban growth in poor countries, and now is the time to plan for the megacities of the future.

Lessons Learned: Transportation

The chapters in Part IV attempt to cover a lot of ground, from the abstract to the practical aspects of policy design. Part IV started with the damage function, which is complex and varies dramatically depending on vehicle, fuel, location, time of day, and weather. The strong interactions between these variables suggest that an integrated approach may be highly appropriate in order to save costs. However, the complexity makes the design of an all-encompassing “first-best” system (such as A in Table 19-4) appear impossible. Although developments in information systems may improve the situation, such information systems are so far used mainly for traffic management (see “K” in Table 19-4), as in Singapore.

TABLE 19-4 A Policy Matrix for Some Environmental Aspects of Transportation

<i>Policy</i>	<i>Fuel</i>	<i>Vehicle</i>	<i>Traffic</i>	<i>Integrated</i>
Regulation	D	H	J	C
Bans	G	N	O	
Price (tax)	B	P	K	A
Information	F	Q	L	
Deposit–refund or two-tiered instruments		I		
Tradable emissions permits	E			
Public provision			M	

Notes: A = environmentally differentiated road pricing (“first-best” system); B = fuel taxes (possibly differentiated); C = urban planning; D = fuel quality regulations; E = marketable permits (e.g., for fossil carbon); F = green labeling of fuels; G = bans on certain fuels or fuel components; H = regulations concerning vehicle emissions, for new vehicles or for inspection and maintenance; I = two-tiered systems that would, for instance, entice cleaner vehicles to “self-reveal” and opt for a certain option in a menu of policies; J = zoning; K = tariffs for public transport; L = green labeling of transportation services; M = physical structures, such as railways; N = retirement, removal, or repair of the most-polluting vehicles; O = restricting traffic in certain zones or at certain times of day or night; P = taxes (or subsidies) on vehicles; and Q = green labeling of vehicles.

Typically, most policies aim to reach subgoals regarding fuel quality, vehicle standards, fuel efficiency, and traffic management and to find suitable instruments for each. Fuel taxes (B) are commonly used. Although not always recognized as—or even intended to be—environmental policy instruments, fuel taxes have a strong effect on fuel use and thereby can be effective in combating global warming. This effect is usually not a top priority in poor countries, but reductions in overall fuel use yield ancillary benefits, such as decreases in local emissions and other externalities. A fuel tax is generally not the best way of achieving all the environmental goals related to the transport sector, but it may be an important part of a package of policy instruments. However, a fuel tax may be regressive in countries such as the United States, where total reliance on the automobile is combined with a fairly unequal income distribution. In countries with more equal income distribution, the issue may be less significant; in many developing countries, fuel taxes may be progressive because poor people have little access to individual motorized transport. The social distribution of the costs of a fuel tax are important for its acceptance. One political aspect of fuel (and other) taxes is the development of more or less powerful lobbies in favor of or against each particular tax.

Fuel quality, vehicle standards, and urban planning are the three most important policy areas for local (urban) environmental management. The choice of policy instruments depends on the context. Urban planning (C) is a regulatory tool and relies heavily on zoning (O) and infrastructure investments (M). Physical regulations and reforms of the pricing and tariff structure are important determinants of demand (e.g., the pricing of public transportation

(K) and, more generally, the organization of markets for bus and other public transportation services).

Fuel quality is a complicated and technical area in which technical standards, bans, and norms are the most common basic instruments of regulation (D). However, experience has shown that many improvements—notably, the transition to unleaded gasoline—have been facilitated by the use of tradable permits (E) in the United States and by differential tax treatment (B) in many other countries. In some cases, information- or reputation-based policies such as green labeling or green marketing (F) or simply bans (G) were used. The choice of instrument appears to depend on the sophistication of the economy, among other things. In some diverse, large economies, the flexibility afforded by market instruments allows cost savings (and preempts political opposition). In smaller economies, the scale economies may tip the balance in favor of simpler instruments to avoid the extra costs of multiple standards and delivery systems. The regulation of new vehicles and particularly vehicle emissions technology (H) is traditionally one of the most central instruments. It also can be supplemented by information and labeling (Q) (see Chapters 17 and 18).

The interactions between fuel and vehicle technology can be intricate. The substitutes available as well as the political dynamics differ by situation (e.g., for diesel, alcohol, and gasoline), because the fuels are used and produced by different categories of business. Sometimes the fuel and vehicle industries try to shift all the blame onto each other (and onto vehicle owners) for problems that essentially have joint causation, raising issues of liability and moral hazard and making it important to have real monitoring and flexibility in the application of policy instruments. For instance, it is not inconceivable that a truck classified as Euro II might, with special ultraclean fuels, or with additional exhaust treatment such as particle filters, meet the emissions requirements that correspond to the next class (Euro III). On the other hand, with poor maintenance and ordinary or poor-quality fuel, its emissions might correspond to an inferior class. If policy instruments are solely based on the vehicle type or the fuel type, then the important interaction between the two will be missed.

Green marketing also has been used in the context of transportation services (L). The electric trams in Gothenburg, Sweden, have an environmental label partly because they run on “green” electricity. Similarly, buses and taxis in Mexico that converted to unleaded gasoline were painted green instead of the traditional yellow.

Because a disproportionately large share of all pollution comes from a few of the worst vehicles, targeted inspection and maintenance might be a powerful instrument (H), but the exact design of such a system is tricky. One question is the degree of homogeneity versus flexibility. Damages due to emissions depend strongly on location and vehicle type. Requiring inspection and maintenance only in cities would be an effective restriction except that it would provide an incentive to register vehicles in rural areas. Flexible systems make monitoring and enforcement considerably more complicated. Introducing

a kind of market-based instrument such as a deposit–refund system (I) might alleviate the monitoring issue by placing the incentives for revelation or signaling with the individual motorist. The idea is to create incentives for the motorists who cause the least environmental damage to reveal themselves through voluntary reporting of some sort.

As shown by such positive examples as Curitiba as well as the many negative counterexamples, conscious long-run planning is a crucial element for improving the urban environment. Such planning includes zoning, infrastructure, institutions, public transport pricing, and choice of technology (J, K, L). The fact that many of these instruments are long-run instruments should in no way be taken as an excuse not to implement them immediately. However, they must be tailored to future technologies, of which knowledge is incomplete. As in Mexico City, combinations of many policy instruments that include both fuel taxation (B) and regulations (H) may be significantly better than either instrument separately. In contrast, partial driving bans (such as “*hoy no circula*”) do not appear to be efficient.

Because technical change is so rapid, policymakers should be skeptical to invest in systems with large and expensive ground-based infrastructure. This applies both to physical structures (such as subways) (M) and infrastructure for monitoring (such as toll stations, if one considers that such systems may soon be out-competed by technically more advanced systems with less fixed infrastructure). Because the development of information technology in general and intelligent traffic systems in particular is rapid, it seems likely that the advantage of more sophisticated systems will increase over time. On the other hand, vehicles also will become less polluting over time as a result of technological progress, tending to somewhat reduce the benefits of this kind of instrument.

Although vehicles and roads have long useful lives and much emphasis must be placed on long-run planning, numerous vital policies must be pursued in the short run, too. Such policies include fuel reformulation (which includes abolishing lead and sulfur); retirement, removal, or repair of the most-polluting vehicles from sensitive areas (N); and removal of perverse subsidies such as tax breaks for older vehicles or subsidies for the manufacture of obsolete vehicle models (P).⁵

The transportation sector presents a range of environmental problems with considerable technical challenges. In some respects, it presents more sociopolitical difficulties than in, say, industry, because monitoring and enforcement are difficult and because the owners and polluters are not professionals, which introduces several political constraints. This area also has experienced remarkable technological progress. Consequently, the cost of additional emissions reductions may frequently be higher (at the margin) than in other sectors. Because mobility is so highly valued and because cars have status, it may be possible to forge ahead with environmental goals faster in the transportation sector than in other sectors. Doing so will require a continued

refinement of policy instruments to ensure that the potential promises of new technology are harnessed.

Supplemental Reading

Bento et al. 2005
Esteban 2007

Fullerton and Gan 2005
West 2004

Notes

- 1 In a speech to the *Automotive News* World Congress, William Clay Ford Jr. (great-grandson of Henry Ford and chairman of Ford Motor Co.) said, “Hybrids and fuel cells will finally end the 100-year reign of the internal combustion engine as the dominant source of power for personal transportation” (cited in Walsh 2000).
- 2 Such ultraclean gasolines are used for chain saws in Sweden (important because the user’s nose is close to the exhaust) and have been suggested for outboard motors in sensitive coastal areas by Swedish marine ecologists (Östermark 1996).
- 3 Some countries only inspect and test for environmental vectors, but testing safety features seems logical, too, because faulty brakes can also cause external damage.
- 4 Economists are skeptical of “free lunches,” but presumably, some uncounted attention cost can explain the apparent paradox of negative cost abatement. In fact, economists are fairly wary of this kind of cost curve estimated solely on the basis of engineering data, but in some cases, they may still be the best models available.
- 5 At least as late as the mid-1990s, Mexico had vehicle tax exemptions for vehicles older than 10 years. Mexico and Brazil long protected their domestic production of vehicles such as the Volkswagen Beetle, which is hardly on the cutting edge of emissions technology.

PART V

Policy Instruments for Industrial Pollution

The sources of pollution associated with road transportation (addressed in Part IV) are small, numerous, and mobile. These characteristics pose several challenges to policymaking, particularly in the area of monitoring.

In contrast, the sources of industrial pollution addressed in Part V are typically large, except perhaps in the informal sector of developing countries. Because polluting firms are usually large and fixed in location, they commonly are prime targets for environmentalists, particularly in the initial phases of environmental awareness. Abatement may be easier to monitor in large industries than in road transportation, but obstacles still exist. Difficulties in monitoring and enforcement stem from the power and influence that industries wield through the economic resources at their disposal as well as the fact that so many people depend on industry for employment, business, and tax receipts (see Chapter 13).

When industries are international, regulations tend to “spill over” from one country to another. Part of the reason may be copying successful legislation, but considerable evidence indicates that multinational industries have a strategy. They initially resist legislation or standards in one country. However, after they have adapted, they proactively promulgate the same standards in other countries to achieve a competitive edge by avoiding the cost of producing for many specifications in different markets. This is one of the many factors

that distinguish policymaking in developed countries (Chapter 20) from that in developing and transitional countries (Chapter 21).

Experience in Developed Countries

In this chapter, we address many topics chosen to illustrate various aspects of instrument design for the industrial sector¹:

- efficiency in the abatement of sulfur emissions,
- abatement of nitrogen emissions,
- green tax reform in Germany,
- prohibition, taxation, and regulation of hazardous solvents,
- global ozone policy, and
- global climate policy.

First, we discuss the energy sector, in which many countries combine regulation with market-based policies to promote energy efficiency, affect the structure of energy supply, or specifically limit emissions (e.g., of sulfur oxides or nitrogen oxides). This discussion provides an opportunity to compare policies proposed for similar goals in different settings. One of the most important issues related to the energy sector is acid rain, and we compare the tax approach commonly used in northern Europe with the permits approach of the United States.

The mainstays of environmental policy still are legislation, liability, and public spending; therefore, we have selected three illustrative case studies. In the first, we compare a ban (Sweden), a tax (Norway), and a stringent regulation (Germany) of trichloroethylene, a particularly noxious solvent. To conclude, we analyze transboundary environmental issues: ozone depletion and global warming.

Abating Sulfur Emissions

Acid rain was one of the first environmental problems to achieve global recognition and has come to symbolize environmental issues in many countries.² It is hardly new; in fact, its effects were recognized almost 2,000 years ago, when copper smelting released large quantities of sulfur, killing vegetation in the Roman Empire.

When Sweden and Norway started to associate the death of lakes and fish with sulfur emissions in the 1960s, the connection was first rejected by Great Britain, Germany, and other sources of large-scale emissions. With the *Waldsterben* (death of forests) in Germany and better research, the connections are well understood today. They are somewhat complicated, because the sensitivity of the underlying rock and soil can vary so much from one country to another. Countries with old rock, like Scandinavia, can be several orders of magnitude more sensitive than countries that have a lot of calcareous rock with a high acid-buffering capacity, like England. The incentives to abate sulfur emissions depend on the perceived environmental damage and vary by country. The response also depends on abatement costs and political factors as well as the treaties that countries are bound by (in this case, mainly the Sulfur Protocol) and the efficiency of the policy instruments they select (e.g., Klaasen 1996).

Table 20-1 shows that several country groups can be distinguished by emissions and rate of emissions reduction. Perhaps not surprisingly, Austria, Norway, Sweden, and Switzerland—countries that are sensitive to acidification—have low per capita emissions and reduced sulfur emissions significantly in the 1980s and 1990s. The United States, Canada, and some southern European countries have larger emissions per capita and have made smaller reductions; Greece and Portugal have even increased sulfur emissions. Emissions in the United Kingdom used to be high but decreased rapidly during the 1990s (mainly as a result of switching from coal to gas). During the past decade, European collaboration has been successful, and many (western) European countries have made considerable efforts to abate. In the transitional economies, emissions have been significantly reduced but usually not as a result of specific sulfur policies; rather, the reductions are due to industries closing down or increased energy efficiency. Many but not all of these countries still have high emissions.

European Policies: Regulation and Tax

Some countries have found “abatement” easy by switching energy sources from oil and coal to gas (United Kingdom) and hydropower and nuclear (Sweden and France), or because large industries were closed (e.g., when East Germany and West Germany merged). Also, the need for energy for heating and the structure of industry and transportation systems are relevant factors that vary considerably between countries. Distinguishing the effect of intentional policies from that of other changes in the economy (e.g., fuel switching, energy savings, and structural change) is tricky; judging the effectiveness or efficiency of the policy instruments used is even more difficult.

Most of the policymaking in this area is regulatory: performance standards (the sulfur content of fuels) or design standards (mandatory technologies for abatement). For example, the European Council’s 1975 resolution on

TABLE 20-1 Sulfur Emissions in Several Countries

Country	Emissions per capita (kg of SO _x), 1997	Reduction (%), 1980–1997 ^b
<i>Low sulfur emitters</i>		
Switzerland	4	78
Austria	7	86
The Netherlands	8	75
Norway	8	78
Sweden	8	86
<i>Medium sulfur emitters</i>		
France ^a	16	73
Germany	18	80
United Kingdom	19	66
Finland	20	83
Belgium	22	74
Denmark	22	76
Italy ^a	23	64
Portugal ^a	37	–40
Ireland	41	26
Greece ^a	49	–36
Spain ^a	49	37
United States ^a	65	26
Canada	91	41
<i>Transitional economies</i>		
Russia	17	66
Lithuania	19	75
Ukraine	23	71
Poland	56	47
Hungary	66	60
Czech Republic	70	69

Notes: ^a1995 or 1996 data. ^bNegative reductions imply an emissions increase.

energy and the environment was intended to reduce the sulfur content in light fuel oil (LFO) and regulate the use of heavy fuel oil (HFO) (Johnson and Corcelle 1995) (see Table 20-2).

In addition to regulating contents, several European countries use differentiated energy or fuel taxation to encourage a reduction in sulfur use; some also use permit trading (see Chapter 6). Sweden, Norway, and Denmark charge a high tax on sulfur emissions (US\$3,000/ton, US\$2,100/ton, and US\$1,300/ton, respectively), whereas taxes in Italy, France, Switzerland, Spain (a regional tax in Galicia), and Finland (only diesel) are all less than US\$50/ton. Although

TABLE 20-2 Sulfur Content Limits in Liquid Fuels (Gas Oil) in European Member States

Article	Came into force	Limit (% by weight)	
		Light fuel oil	Heavy fuel oil
75/716/EEC	Before 1980	0.5	0.8
75/716/EEC	Oct. 1, 1980	0.3	0.5
87/219/EEC	Jan. 1, 1989	0.3	
93/12/EEC	Oct. 1, 1994	0.2	
93/12/EEC	Oct. 1, 1996	0.05	

Source: Ercmann 1996.

all taxes should, in principle, have some effect, the effect of very low taxes is unlikely to be significant. Even in the high-tax countries, the effect is hard to separate from the effects of concurrent policies. In Sweden, the sulfur emissions tax was estimated to be responsible for 30% of the reduction between 1989 and 1995 (SEPA 1997).³ The decline in oil use was the main factor behind decreasing sulfur emissions in Swedish manufacturing, and this decline depends on the general policy of high fuel taxation combined with a rapid expansion in nuclear power during the same period (Hammar and Löfgren 2001). However, the sulfur tax also appears to have been important in reducing the actual sulfur content of fuel oil below the legal limits. For instance, the limit for LFO was 0.2% in 1976, but actual sulfur content was lower. In 1991, the actual sulfur content was 0.08%, and in 1994 it was 0.058% when the legal limit was still 0.2%; this result must be mainly a result of the tax.

Poland deserves special mention in a discussion of sulfur taxation because among eastern European countries, Poland has most consistently applied tough economic instruments to reduce pollution and to finance the abatement of pollution; the fees are paid into funds. Poland's sulfur tax was between US\$60/ton and US\$100/ton between 1990 and 1996 (Klarer, McNicholas, and Knaus 1999)—although low by Scandinavian standards, it was higher than that of any other OECD countries. Considering the nation's low average income and high overall pollution levels, this tax is sizable (comparable to sulfur permit prices in the United States). The emissions levels may not have declined as quickly in Poland as in other formerly planned economies, but to a large extent, the smaller amount of de-industrialization is responsible. The tax is believed to have been successful, through both its incentive effect and the funds for abatement investments that it generates.

U.S. Policies: Tradable Permits

The other major sulfur policy experiment is the U.S. trading scheme. Title IV of the 1990 Clean Air Act Amendments (CAAA) established a market for tradable

sulfur dioxide (SO₂) among electric utilities.⁴ Title IV was the first large-scale, long-term environmental program in the United States to rely on tradable emissions permits. Although the trading of abatement credits had been allowed in numerous earlier programs, it typically had been an auxiliary mechanism, whereas this time, it was the primary mechanism. Earlier air pollution regulations in the United States (and many other countries) targeted individual sources and their emissions rates. In this program, the focus was on the aggregate emissions, which were capped (although only from the electricity sector).

The legislation was intended to reduce emissions by roughly half, that is, by 10 million tons per year. (Total U.S. emissions in 1990 were about 21 million tons, of which 17 million tons were emitted from the electricity sector.) To achieve this goal, permits equivalent to about 9 million tons were issued.⁵ The reductions are targeted to the electricity sector, and all fossil-fueled plants are obliged to hold several permits that correspond to their emissions. At the end of the year, utilities must prove their compliance: the allowances in their accounts with the U.S. Environmental Protection Agency (U.S. EPA) Allowance Tracking System must equal or exceed annual SO₂ emissions. There is a 60-day grace period to purchase additional SO₂ allowances. Remaining allowances may be sold or banked for future use. Trading is unrestricted. Individual plants decide how much to abate and how many permits to buy, and anyone can buy permits—utilities, fuel suppliers (who may, for instance, bundle the sale of coal with that of permits), environmentalists (who might want to reduce aggregate pollution by holding but not using permits), and brokers speculating in rising permit prices.

When the CAAA sulfur reduction was being planned, the main avenue for reductions was thought to be scrubbing and other technological forms of abatement. Estimates of abatement costs varied considerably; values of US\$250–\$350 for Phase 1 (from 1995) and US\$500–\$700 for Phase 2 (starting in 2000) were calculated (Joskow, Schmalensee, and Bailey 1998). The early (1990) estimate from U.S. EPA was US\$750, and pre-1989 industry estimates were US\$1,500 for emissions permits, which was also the U.S. EPA price for direct sales (Burtraw 1998a, 1998b). Another upper limit is the penalty payable for failure to hold permits, which is US\$2,000/ton. Yet other estimates are illustrated in Figure 20-1.

One of the reasons for choosing this design was that earlier policies had some undesirable features. They targeted the new sources more than the old ones (referred to as “anti-new-source bias”) and required similar technologies or emissions reductions of all plants without properly accounting for the characteristics of different kinds of coal. Because abatement costs appeared to be considerably heterogeneous, the potential efficiency gains from a market-type policy instrument were expected to be large (see Chapter 9). As it happens, the main variation in cost has been between western producers of “cleaner” low-sulfur coal and eastern high-sulfur coal mines. CAAA required power plants in the west to include scrubbers on new plants that operated with fairly clean

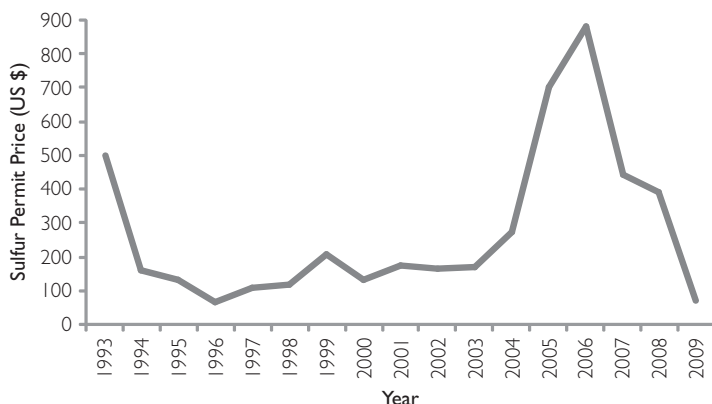


FIGURE 20-1 Sulfur Permit Prices

Notes: Value for 1993 is an approximate average for pretrading predictions, which ranged from \$309 to \$981. Prices correspond to the weighted average of spot bids. *Source:* U.S. EPA www.epa.gov/airmarket/trading/auction.html

coal, whereas old eastern plants could continue, with fairly limited regulations, to burn high-sulfur coal. This policy was both unfair and inefficient.

The program coincided with deregulation of the railways and provided considerable impetus to the transport and marketing of low-sulfur coal across the United States. Freight rates dropped dramatically, and the high-sulfur coal has been almost outcompeted by the cleaner coal from the western states. As a result, power stations have been able to meet a large share of their desired reductions cheaply by switching coal suppliers instead of investing in technological improvements such as scrubbing. The price of permits has been much lower than forecasted (see Figure 20-1), although it has increased sharply during some periods due to uncertainties regarding the introduction of further SO₂ regulations and increased permit price expectations. For example, in 2005 the U.S. EPA promulgated the Clean Air Interstate Rule, a cap-and-trade program that will reduce dramatically the SO₂ emissions that move across the boundaries in 28 eastern states and the District of Columbia from 2010 onwards.

The fact that the price of permits has been much lower than expected is often taken as evidence that permit trading is an efficient instrument, and although it is somewhat true, the rapid and large fall in permit price is a reflection of “technological development” or “substitution” that has been unusually successful. The brilliance of the policy instrument would be if this development were due uniquely to the trading, but at least part of this fuel switch might have occurred under other policy instruments. Several observers note that the reduction in permit price was largely a result of reduced freight rates, extra allocations of permits, and local state rules that provided an extra constraint in some cases, leading to large abatement investments (Ellerman and Montero 1998; Burtraw 1998a, 1998b; Joskow and Schmalensee 1998;

Joskow, Schmalensee, and Bailey 1998; Burtraw and Palmer 2003; Chestnut and Mills 2005; and Burtraw and Szambelan 2009). However, the cost savings stemmed from tradable permits rather than regulatory rules, which “could have made abatement more than twice as expensive” (Fullerton, McDermott, and Caultkins 1997). This example illustrates the importance of using general policy instruments that allow economy-wide processes to work instead of targeting narrow technological solutions.

Sulfur emissions exhibit heterogeneity in both abatement and damage costs. As discussed in Chapter 9, market-based instruments have considerable potential to reduce costs when abatement costs are heterogeneous. However, when damage itself is heterogeneous, market-based instruments are less suitable, and instruments such as zoning have an advantage. In the case of permit trading to reduce sulfur, the relevant risk is that health damages might increase if the patterns of emissions change unfavorably. More detailed models show that trading and banking led to sizable shifts in the geographical (and temporal) distribution of emissions—in some states, effects as large as 20% compared with a baseline that was half the original levels (Burtraw and Mansur 1999).

The overall effects of these shifts are beneficial, not detrimental, in terms of total health costs because emissions appear to be moved away from the most densely populated regions. This effect is partly due to regulatory pressure on abatement; allegedly, “downwind” states (such as New York) also pressure their utilities into not trading with “upwind” states (such as Pennsylvania and Ohio). Banking led to more rapid decreases in the 1990s (and thereby increased future emissions). The aggregate cost savings from trading and banking are estimated to be 13% of compliance costs in 1995 and 37% in 2005. Potential annual cost savings due to trading are US\$800 million (1995 dollars) or 43% of compliance costs under an “enlightened” command-and-control program in which the regulator adopts a performance standard approach (Carlson et al. 2000). The savings would be even greater when compared with a less enlightened command-and-control program with uniform technology standards.

Permit allocation has received a great deal of attention in the literature. The expected value of the permits at the onset of the program was around US\$5 billion/year, and thus lobbyists were prepared to fight for every detail in the allocation (Joskow and Schmalensee 1998). The main mechanism used for determining permit allocation in the United States has been somewhere between grandfathering and output (or rather input) allocation (see Chapter 6). Permits have been free and allocated based on historic values—not in proportion to historic emissions but past values of heat input.

An allocation based on emissions (traditional grandfathering) would have been somewhat more beneficial to the “dirtier” plants, whereas an allocation based on electricity produced would have favored the more thermally efficient plants. In actual policy, important issues that affect the decisions include the timing of allocations, the lifetime of the permits, the variables and data on which to base decisions, the choice of base year, and the categorization of

plants that have been built since or were closed (part of) the base year, among others. One of the vital questions, for instance, is whether to have permanent allocations or to update them. Lobbyists from the affected firms and politicians from the concerned towns or states find all kinds of arguments in favor of one or the other principle or additional allocation rule. Presumably, the decisionmakers settled on permanent allocation to avoid repeated opportunities for manipulation. (Allocations are for 30 years but are automatically updated, making them effectively quite permanent.)

Because this legislation had to pass a Congress that was fairly skeptical to environmental legislation, it met with many pitfalls. For the proponents of the law, it was vital to secure support wherever it could be secured, even if it meant compromising somewhat on permit allocation rules. One might think that the political allocation process would severely skew the result. However, although the high-sulfur-producing states were able to impose considerable (abatement and environmental) costs on the rest of the country in the regulations of the 1970 Clean Air Act and its 1977 amendments, they generally were not successful in getting a disproportionate share of allowances in the more market-based revision of 1990 (Joskow and Schmalensee 1998).

In one positive summary of this policy experience, U.S. sulfur trading is portrayed as an efficient instrument along the lines of those illustrated in economics textbooks: the “environmental goals were achieved on time, without extensive litigation and at costs lower than projected” (Joskow and Schmalensee 1998). One point essential to this efficiency is ensuring that the permit market really develops; in this case, U.S. EPA held annual auctions for 2.8% of the permits that had been withheld (that would otherwise have been allocated to individual units). This sale was revenue-neutral, because the proceeds of the auctions were handed over to the plants that would otherwise have received these permits. The auctions helped start the process of price discovery by the participants in the market. They also helped allay the fear that the initial permit recipients would “hold on to them” and that trading would thus be insufficient to satisfy the needs of new sources.

The sulfur-trading program is a big step ahead in mechanism design. Future programs may develop even more efficient mechanisms and will have to adapt to the characteristics of individual pollutants. It is difficult to say whether taxes are preferable to permits. As discussed in Chapter 11, taxes (and auctioned permits) have output and revenue-recycling effects that free permits do not have. Presumably, the biggest difference is in political acceptability, and the best-suited instrument may depend on the pollutant and the political setting. For the extent of sulfur reduction, the most crucial factor is probably not the choice of policy instrument but the effective tax level (or number of permits and thus permit price). Looking back at the unexpectedly low permit price, some observers consider that the number of permits was too high and the goal of 50% sulfur reduction much too modest (see Table 20-1). It may partly be a reflection of U.S. priorities; acid rain is not

as high a priority as it is in Scandinavia. However, for the purposes of this discussion, it would be interesting if something in the mechanism choice biased the outcome. Grandfathering of permits is a fairly polluter-friendly policy. Other policies, such as taxation, probably would not have had a realistic chance of being implemented at this level of effective control. However, it is worth considering what a step-shaped function (such as that suggested in Requate and Unold 2001) could have achieved. For sulfur emissions, such a mechanism might have resulted in a smaller (free) allocation of permits combined with a menu of call options (or auctions) for additional permits (see Chapter 10).

Reducing NO_x Emissions from Combustion

Nitrogen oxides (NO_x) are the other major precursor of acid rain.⁶ Chemical effects of NO_x are not the same as those of SO₂, but they are perhaps more comparable for these two pollutants than for many other pairs. Several technical differences are relevant for their abatement costs, monitoring, and the politics of instrument choice. In general, NO_x emissions reduction has been less successful than that of SO₂, largely because of inherent technical difficulties. Most of the NO_x is formed by the effect of high temperature on atmospheric nitrogen. Unlike sulfur, NO_x does not come from an impurity in the fuel. As a result, NO_x emissions cannot be easily predicted, and complicated, expensive monitoring equipment is required.

Refunded Emissions Payments in Sweden

For geological reasons, Scandinavia is especially vulnerable to acidification; most of the area is on granite bedrock. Consequently, nitrogen oxides and sulfur oxides have been important environmental policy targets in Sweden and Norway. Even so, the NO_x emissions reduction in Sweden between 1980 and 1995 was approximately 20%, whereas emissions of SO₂ were reduced by 80%. Part of this difference is related to the complexity of NO_x formation and monitoring, which has made it impossible to impose a broad-based tax on all NO_x. Instead, the Swedish Parliament decided in 1990 to introduce a charge of 40 SKr per kilogram of NO_x emitted from all combustion plants that produce at least 50 gigawatt-hours (GWh) of useful energy per year. The charge came into effect in January 1992 and affected about 200 plants. The level of the NO_x charge (roughly US\$4,000/ton) was determined by a Swedish Environmental Protection Agency (SEPA) study in 1987.⁷

The design of the charge is unique. On the one hand, it is high—more than 200 times higher than the French charge and much higher than U.S. permit prices. On the other, its proceeds are returned to the polluting companies, relative to the amount of energy produced by the specific plant. This refunded

emissions payment (REP) scheme (see Chapter 7) means that the polluting industry as a whole does not pay anything to society—presumably, this fact has made the charge politically feasible. In fact the industrial lobby is split and its power reduced considerably since about half the firms (the cleaner half) actually make more money from the refund than they pay on the fee (Fredriksson and Sterner 2005). The design mechanism was chosen partly because only large combustion plants are obliged to pay the charge. This decision was based in part on the high costs of metering, which (together with abatement costs) were considered unreasonable for smaller plants. If a tax were applied to only a subsection of an industry, then that subsection would be unfairly disadvantaged compared with other firms in the same industry. If the tax were applied only to the large plants, then companies such as Gothenburg Energy would have an incentive to set up several small combustion plants instead of one large one, and this response typically is not desirable in any respect, including the emissions of NO_x and other pollutants.

As the system has developed and because it appears to be effective, costs for abatement and metering have decreased, and the criterion for inclusion has been lowered twice. In 1996, plants producing at least 40 GWh of useful energy per year were included, and in 1997, the level was lowered to 25 GWh. In 1998, 400 plants were subject to the charge, compared with 200 plants in 1992.

Tax Levels, Use of Tax Revenues, and Administrative Costs

In a REP scheme, the cleaner-than-average firms make a net gain (which may or may not be larger than abatement costs), whereas the rest have to make a net payment. Total revenue from the Swedish NO_x charge (again, refunded to the plants) was the equivalent of between US\$50 million and US\$100 million between 1992 and 1998 (SEPA 1997; Naturvårdsverket 2002).

The charge is administered by SEPA, and administrative costs have been kept low—to approximately 0.3% of revenues collected. Metering costs are estimated to be approximately 3% of total charges paid. NO_x emissions are measured hourly by sophisticated SEPA-approved equipment. If a firm cannot measure its emissions temporarily due to technical problems, a high emissions factor is assumed for that period; therefore, most firms have a very high rate of compliance. All plants are obliged to submit a form with information regarding their production and NO_x emitted. SEPA audits the firms and randomly selects several firms each year for detailed review. The goal is to review each firm on average every fifth year. At the beginning of each calendar year, each plant submits a declaration of its emissions and energy produced for the previous year. Charges are due before October 1, and SEPA refunds the plants before December 1 of the same year. Each firm can have several production units monitored separately.

Environmental Effectiveness, Efficiency, and Links to Other Instruments

The NO_x charge is generally thought to have been rather successful. Firms adapted quickly to this economic policy instrument, and emissions reductions started immediately (see Table 20-3). The positive environmental effect was biggest in the beginning, but improvements have continued, although at a decreasing rate. This reduction has been achieved despite the enlargement of the scheme in 1996 and 1997.

The policy has brought considerable technological progress in lowering emission intensities. Not only the larger plants have made rapid progress in emission reductions; the smaller units, with initially higher emission intensities, have also reduced their emission intensities significantly. There has been both a great deal of innovation and of technology diffusion. In addition to major investments in new technology and specific abatement technologies,

TABLE 20-3 Summary Statistics Regarding Combustion Plants Subject to NO_x Charge in Sweden, 1992–2008

<i>Year</i>	<i>No. of plants</i>	<i>Energy produced (GWh)</i>	<i>NO_x emissions (tons)</i>	<i>Emissions coefficient</i>
1992	182	37465	15305	0.41
1993	190	41158	13333	0.32
1994	203	45193	13025	0.29
1995	210	46627	12517	0.27
1996	274	57150	16083	0.28
1997	371	54911	15107	0.28
1998	374	56367	14617	0.26
1999	375	54921	14050	0.26
2000	364	51399	12765	0.25
2001	393	58142	14160	0.24
2002	393	61014	14730	0.24
2003	414	66136	15836	0.24
2004	405	65758	14930	0.23
2005	411	64812	14374	0.22
2006	427	67284	14513	0.22
2007	415	69152	13590	0.20
2008	408	70111	13568	0.19

Note: Emissions coefficient is measured in kilograms of NO_x per megawatt-hour of energy produced.

Source: Elaborated from data provided by SEPA.

learning by doing and interactions with other environmental and social goals have played important roles (see Sterner and Turnheim 2009).

In addition to the NO_x charge, many plants are subject to local regulations. Hence, observers argue that the reduction in NO_x emissions is due to both the charge and local standards. According to one study, approximately two thirds of the reduction can be attributed to the charge and the rest to the local regulations (SEPA 2000). The method for arriving at this estimate is shaky, however, and almost all the plants seem to be operating below their legal requirements. As a result, the charge alone might have given almost the same effect without the standards, which makes it difficult to allocate shares between the two instruments.

The dual advantage of a REP scheme instead of an “ordinary” tax is that the incentive for reducing emissions is strong while the liable firms are not subjected to a heavy economic burden. The fact that the net payment for the companies is smaller than a tax (and zero on average) has made this instrument acceptable to firms. In the United States, firms have considered grandfathered tradable permits preferable to taxes because they place a smaller burden on firms and are more neutral regarding competitiveness. The Swedish NO_x example shows that it is possible to create a financially neutral tax as an alternative to tradable permits.

One potential drawback with refunding the charge is that the instrument thereby does not satisfy the more stringent interpretation of the polluter pays principle and therefore does not give an optimal resource allocation (i.e., there is no output or revenue-recycling effect). An ordinary environmental tax (or permit scheme) would increase total and average production cost, hence raising the market price of the final good. The changed price due to internalizing the environmental damage in the production cost gives signals to the economy that would, in the long run, induce structural changes. In an economic market model, these price responses are part of the optimal adjustment. They are not as strong with a refundable charge; the product price will rise because of abatement costs but not because of tax payments. From a general equilibrium viewpoint, this limited price response is unfortunate. However, for many participants in the economy, it may be seen as an advantage: production, employment, and so forth are not cut back; prices do not rise (as much as with a tax). In fact, NO_x might be reduced not by decreasing overall consumption and production of energy but only through technological abatement. In such a case, the refunded charge would be the ideal mechanism.

Furthermore, refunding makes it feasible to set the charge at a relatively high level. A tax of the same magnitude would meet enormous resistance from the entire polluting industry, whereas the refunded charge is a benefit to the cleaner companies. It has the effect of splitting the industry lobby. Finally, the effect on prices depends on what it is measured against. Compared with doing nothing, price increases due to the cost of abatement. It is still a small increase that reflects abatement costs—roughly equivalent to the increase if (optimal)

regulations had been used to achieve the same reduction. Compared with the use of taxes of the same magnitude, product price increases less.

Policies in France, Spain, Italy, and the United States

In 1985, the French government started to apply the polluter pays principle to tax SO_2 emissions. This law was renewed and expanded several times to incorporate several other air pollutants, including NO_x . In 1998, the rates were increased to the equivalent of US\$40/ton of NO_x or volatile organic compounds (VOCs) (and US\$25/ton of SO_x or HCl). One assessment of the tax estimates that NO_x emissions have been reduced by 6% since the mid-1990s (ADEME 1998). This reduction can hardly be due to the direct incentive effects of the tax, because the tax is low and based on estimated emissions rather than actual emissions. However, the proceeds of the tax are used partly to give grants for abatement technology and research, and the latter are assumed to have led to emissions reductions.

Italy has a tax on NO_x of around US\$100/ton, which is higher than that of Spain and France but not Sweden. The tax applies only to large combustion plants (>50 megawatts/year) and to emissions that exceed the standards specified in the EU Large Combustion Plant Directive. It seems to suggest that it is applied only in special cases as a form of penalty. Similarly, the autonomous region of Galicia, Spain, taxes NO_x at the equivalent of about US\$30/ton (Bokobo Moiche 2000).

In the United States, NO_x is not considered a problem related to acid rain but primarily as a precursor to ground-level ozone. Because the severity of environmental effects varies considerably, NO_x is subject to several distinct abatement programs differing from state to state. Title IV of the 1990 CAAA specifies a two-part strategy to reduce NO_x emissions from coal-fired power plants: reduce annual NO_x emissions in the United States by 400,000 tons per year between 1996 and 1999 (Phase 1), and by approximately 1.2 million tons per year beginning in 2000 (Phase 2). These reductions correspond to less than 5% of total emissions and, according to U.S. EPA, translate to abatement costs of a few hundred dollars per ton. Essentially a traditional regulatory program, the program does contain some provisions for optional trading. It allows utilities to “average” emissions over separate units but not to bank or trade with other utilities.

A couple of other U.S. programs also have NO_x targets. One is RECLAIM (see Chapter 6), an ambitious program that aims for an 80% reduction of NO_x (and SO_x) from 1994 to 2003 in the south coast region of California. It is not completely a cap-and-trade program but rather a conventional “bubble” trading program. Each licensed source has to reduce its overall emissions (from a whole facility), and if a firm overcomplies, it can trade the excess permits.

A regional U.S. emissions trading program known as NO_x SIP Call targeted only summertime NO_x emissions in several eastern states. Its structure was

more in line with the cap-and-trade sulfur emissions program, and from the beginning of its implementation in 2003 to 2008, it dramatically reduced NO_x emissions from power plants and industrial sources during the summer months in 20 eastern states and the District of Columbia); in the 2008 ozone season NO_x emissions were 62% lower than in 2000. In 2009, however, the program was replaced by the Clean Air Interstate Rule (CAIR) NO_x ozone season trading program (U.S. EPA 2008).

The allocation mechanisms, which differed from state to state, was the subject of much debate. Many states used a form of output allocation instead of the usual grandfathering or fairly frequent updating of allocations. This made this so-called Ozone Transport Commission (OTC) program more similar to the Swedish REP policy for NO_x (and to the U.S. EPA Lead in Gasoline Phaseout program).

Table 20-4 summarizes the main differences among the instruments used in these four countries plus Sweden (in addition to local standards used in each country).⁸ Perhaps the most striking difference is the level of the charge itself, which is highest in Sweden. It presumably was only politically possible because of the special construction of the policy with refunding. It is somewhat difficult to compare the results analytically because Sweden has a much higher degree of detailed monitoring, as is expected with such a high charge level.

A high charge is probably the only really effective way of achieving a sizable reduction. In five or six years, the Swedish system verified NO_x reduction of more than 30%, starting from a level that was low already because the country had been pursuing an aggressive policy on the precursors of acid rain for many years. Lower tax levels would have less of an effect, but using the revenues to subsidize abatement equipment (as in the French system) should also reduce emissions. However, in Sweden, continuous monitoring

TABLE 20-4 Comparison of Policy Instruments for NO_x Phaseout

<i>Criterion</i>	<i>Sweden</i>	<i>France</i>	<i>United States</i>	<i>Italy</i>	<i>Galicia, Spain</i>
Charge level or permit price (US\$/ton)	5,000	40	200–500	100	30
Pollutants covered	NO _x only (taxes for SO _x)	NO _x , SO _x , HCl, VOCs	NO _x	NO _x	NO _x , SO _x
Instrument	Refunded charge	Charge	Tradable permits	Fee for excess pollution	Tax
Use of funds or allocation of permits	Returned to companies	Abatement, research, etc.	To companies		General (5% for environmental restoration)

Notes: NO_x = nitrogen oxides; SO_x = sulfur oxides; HCl = hydrochloric acid; VOCs = volatile organic compounds.

indicated that the mere installation of equipment generally did not have as great an effect on emissions as it should according to engineering projections. To realize the potential emissions reductions, companies had to experiment with trimming and operational adjustments, which can be done only if there is continuous-time monitoring and if plant operators can strive for incentives (such as permits or charges based on actual measured emissions).

Green Tax Reform in Germany

Since the early 1990s, several European countries have introduced comprehensive green tax reforms. The idea is to use environmental or resource taxes that are needed anyway to finance a reduction in other (more distortive) taxes. Although this idea has merit in some cases, it has been oversold and heavily criticized (see Chapter 11 on double dividends). A constant tax burden seems an essential condition for the acceptability of environmental taxes. Nevertheless, although environmental taxes are powerful instruments for solving environmental issues, they do not necessarily generate funds that are large enough to have a significant impact on the overall budget or on other taxes.

Germany initiated a green tax reform in April 1999. Its main goals were to stimulate energy savings—in the context of the German objective, to reduce CO₂ emissions—and to increase employment. The German reform was implemented through the introduction of two laws, which endorsed five energy tax increases over the period 1999–2003 (Agnolucci 2009). In the first step, uniform tax rates on the consumption of heating oil and road fuels were introduced. An electricity tax was also launched, while the existing taxes on light heating oil and natural gas were increased. The subsequent four steps included increases in the taxes on electricity and road fuels. The increased tax burden was compensated for by reduced social security contributions for both employers and employees.

A number of special provisions and exemptions apply to different energy sources—particularly renewable sources, and to some economic sectors. Estimates indicate that the environmental green reform has been successful in reducing energy and CO₂ emissions while leaving employment levels qualitatively unchanged (Agnolucci 2009).

Evidence has also shown that the implementation of environmental taxes is subject to a number of difficulties. The main obstacles are related to the distributive and competitiveness implications. Indeed, because there is some resistance to EU environmental legislation (particularly from the “cohesion countries” of southern Europe), the more environmentally proactive countries make unilateral decisions. This behavior raises issues of competitiveness, particularly in small open economies. It also was evident in the German government SPD/Grünen (Social Democrat/Green) coalition proposal for ecological tax reform of November 1998, which had a long, detailed list of

industries suggested for exemption from the new tax (Deutscher Bundestag 1998, 5). This list included several kinds of glass, cement, steel, aluminum, fertilizer, and other energy-intensive industries. These industries are the most affected—but they are also the ones that *need* to be affected, both in their choice of technology and their contribution to the overall output mix of the economy, if the policy is to have the intended effect. Exceptions of this kind go strongly against the grain of current thinking in public policy, which tries to promote transparency and homogeneity of rules. In the final law approved in 1999, the exceptions were not included. Instead, the proposed tax increases were reduced heavily for all industries, which in fact pay only 20% of the new ecological energy taxes borne by others (e.g., households). One of the most contentious areas for the German tax was electricity taxation. The general tax on electricity is now around US\$10/MWh. Originally, this tax was going to be differentiated by type of electricity production, to give preference to ecologically sustainable means of producing electricity. This proposal ran into problems related to trade deregulation within the European Union. Legislation does not allow taxes on imported electricity to be higher than the lowest of the applicable taxes within the country. This law makes it almost impossible to differentiate electricity taxes by source of production. Because the European electricity net is becoming both physically and legally fully integrated, it appears to make any national environmental taxation of electric energy difficult. Taxing inputs such as coal or oil would have been an alternative at the national level, but technically this means that the direct national tax on electricity is zero. As a result, imported electricity also has to be free of tax, even if it is produced in a polluting way. If this pollution is transboundary, the importing country may be affected by the pollution but unable to do anything about it.

Another interesting case is the KohlePfennig, a direct cross subsidy to the coal industry abolished in 1997. Prior to 1997, a dedicated electricity fee had subsidized domestic coal production. (In addition, regulations required German power and steel producers to burn a minimum share of German coal.) The coal industry fee was abolished because the German constitution does not allow this kind of earmarking. Taxes are to be for general purposes, and the Bundestag must be able to assign money as it pleases. Instead, direct, overt subsidies are now granted to the coal industry each year.

Much remains to be done, because the tax systems are far from satisfactory with respect to the environmental parameters involved. Many environmental aspects are complicated, and the exact environmental damage depends on many local details. Other problems are global (or regional), and thus national taxes are the wrong level of action. Reform meets considerable resistance in the form of legal issues and special interests of energy suppliers, customers, and trade (and other) organizations.

Prohibition Compared with Other Policies: Trichloroethylene

Chlorinated (or halogenated) hydrocarbons are good solvents for degreasing and dry cleaning; they are chemically active and dissolve fats (including those that protect nerve cells).⁹ These same properties have made these chemicals controversial since environmental problems and health hazards were discovered. Many chlorinated or halogenated hydrocarbon compounds are hazardous or toxic directly or indirectly (e.g., they may form extremely toxic dioxins when burned)—for example, the effect of chlorofluorocarbons (CFCs) on the ozone layer. In this section, we focus on prohibition as a policy instrument in Sweden for one of these solvents, trichloroethylene (TCE), and briefly compare it to the policies followed in two other countries: in Norway, TCE use is taxed; and in Germany, TCE use is strictly regulated.

Degreasing is common in the metals industry. As soon as an object is processed, surfaces must be greased to prevent corrosion. The grease must then be removed for later steps such as welding, lacquering, assembly, and delivery. Health and environmental issues related to these chemicals are complicated, and priorities change with knowledge. This changing nature poses challenges for the design of policy instruments. For some time before damage to the ozone layer was discovered, several ozone-depleting substances (ODSs) such as CFCs were introduced as substitutes for other kinds of chlorinated solvents because they were less hazardous to human health. After the phaseout of CFCs to protect the ozone layer, there was some substitution back to TCE, perchloroethylene (PER), and methylene chloride. Other alternatives to chlorinated solvents in metal degreasing are water-alkaline processes and low-aromatic mineral oils.

Sweden

PER is chemically close to TCE and is used mainly in dry cleaning. The only regulation of PER in Sweden is a tough maximum exposure limit in the working environment of 10 ppm. Although PER is not banned, its use has decreased by 85% from 1988 to 1995 mainly because of a transition in dry cleaning from open to closed systems. However, some open systems still exist, and, according to SEPA (1997), many are poorly maintained and have high emissions. Only 20% of the open machines have carbon filters, and significant low-cost reductions could be achieved (23–47 SKr/kg). Because many dry cleaners are located in apartment buildings, the risk of unintended exposure is relatively high. Paradoxically, the use of TCE is prohibited, but not PER.

In 1991, the Swedish Parliament passed a law prohibiting the professional use of TCE and methylene chloride, effective Jan. 1, 1996. The use of TCE in consumer products was banned starting in 1993. A complete prohibition on all use appears to be a strong policy instrument but has not been wholly effective. The strength of the instrument is in some sense its weakness: the ban creates a strong opposition among users who find it particularly difficult

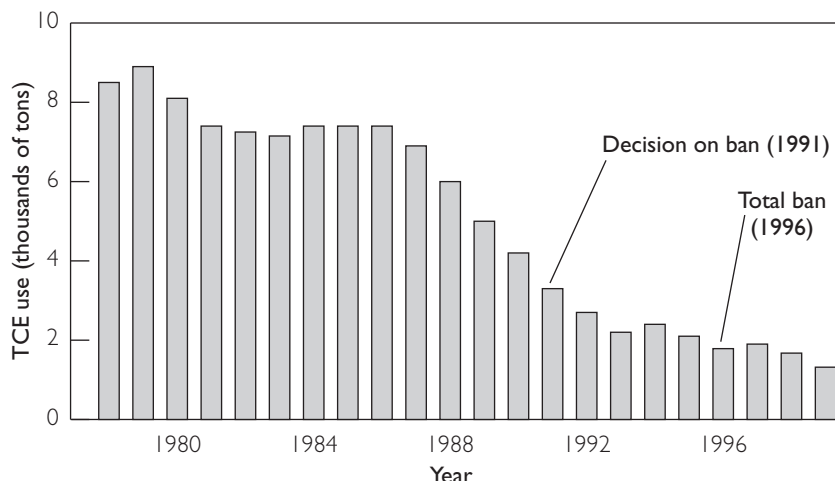


FIGURE 20-2 Use of TCE in Sweden, 1978–1999

Source: Naturvårdsverket (the Swedish Environmental Protection Agency).

to replace TCE or who simply disapprove of the timing or policy method. The ban also has been controversial because the evidence for TCE's toxicity or environmental hazards is less clear-cut than for some other solvents. TCE was sometimes preferred because it was considered less hazardous than other alternatives, such as CFCs and highly aromatic petroleum products. In the early 1980s, the SKF factory in Gothenburg used 3,000–4,000 tons of TCE per year—half of Sweden's total consumption—as a replacement for CFCs and highly aromatic petroleum products.

Figure 20-2 shows the decline in TCE use in Sweden up to 1999, three years after its prohibition. By the time the ban was decided, consumption already had been reduced from almost 9,000 tons/year to around 3,000 tons/year. This reduction may partly have been an effect of the ban, because industry could have observed the process of legislation and decreased TCE use in anticipation of its prohibition. On the other hand, several other countries experienced similar decreases in use. During the late 1980s, large industries were faced with tight emissions standards, which led them to replace technologies and solvents, or to adopt closed systems and carbon filters.

SKF is an interesting case study not only because of the company's size and significant use of solvents but also because its precision demands are unusually exacting. When removing water droplets from ball bearing components, SKF requires the elimination of all water within 30 seconds; after that time, corrosion becomes unacceptable. If SKF can eliminate the use of TCE, then most other industries should find it possible. Reduction in TCE use at SKF was driven by two factors: trade union demands for a safe working environment, and SEPA requirements for reduced emissions. SKF was ordered to

reduce emissions from 250 tons/year to 15 tons/year and to quickly install active carbon filters. The installation alone led to a two thirds reduction, down to about 80 tons/year. Additional reductions required process changes. A new degreasing process using water and low-aromatic oils was installed; new packaging and storage routines were combined with the use of lighter oils instead of wax for the conservation of ball bearings. Finally, only small amounts of TCE were used, so the costs of maintaining the handling, storage, and filter facilities became disproportionately high. Also, SKF saw potential benefits in its environmental image by being proactive (instead of reacting passively to SEPA instructions) and in standardizing processes across facilities. The company therefore decided to phase out all use of TCE in all SKF plants, in Sweden and abroad, even if local authorities did not require it. This decision is one example of the international spillover of environmental standards.

The Swedish ban on TCE also led to bitter opposition and protest; petitions were written, and several companies fought the legislation in the courts. We summarize only some of the more salient points here. In 1994, 39 firms wrote an open letter to the prime minister, saying that the prohibition was poorly motivated and should be withdrawn. They went so far as to say that their competitiveness was threatened and that they would have to move abroad if the prohibition were enforced. In later interviews, several company representatives admitted that the points had been exaggerated. Most of the companies eventually did phase out TCE use, and none of them moved abroad. However, the tone of the letter reflected their strong resentment.

In the first year (1996), some 500 companies were given waivers; however, after that, only companies that could show they were making serious efforts to substitute for TCE were to be granted exemption. Exempted firms also were charged an exemption fee of 150 SKr/kg, which was not only an environmental tax but intended to remove any unfair disadvantage that a complying company would suffer vis-à-vis its noncomplying competitors. Hundreds of companies continued to apply for exemption; when it was rejected, they appealed the decision. The Stockholm court reversed the decision by SEPA, and several additional rounds of appeals progressed to higher courts.

The Netherlands, the United Kingdom, and the EU Commission also started to criticize the Swedish ban. Sweden had no domestic production of TCE (it was all imported), and the ban was thus a barrier to trade in this commodity. The case of one company, Toolex, was referred by a Swedish court to the European Court of Justice (case C-473/98) to determine whether Swedish prohibition was in accordance with the free movement of goods. Meanwhile, the Chemical Inspectorate (*Kemikalieinspektionen*) modified its rules for exemption, dropping the requirement of a phaseout plan and removing the fee, which the EU Commission considered “out of proportion” to the environmental damage. The only requirement left for exemption was that the company actively seek other alternatives and avoid harmful exposure. Several interesting principles of European law underlie this case. On July 11, 2000, the court ruled that the

Swedish prohibition is not counter to EU legislation on the free movement of goods. The reasoning behind the ruling was as follows:

- The basis for the prohibition is concern for health and environment.
- The European Union has classified TCE as toxic and carcinogenic.
- Member countries have the right to stricter environmental legislation.¹⁰
- There is no basis to assume the prohibition was motivated by an attempt to stop trade.
- There are reasonable possibilities of obtaining a waiver.

Hence, one might conclude, with some exaggeration, that the prohibition was accepted because it was watered down by more generous standards for obtaining waivers.

To discuss alternative policy instruments, the structure of the abatement costs must be known. Based on applications for exemption as well as interviews, estimates for the entire TCE-using industry were compiled (Slunge and Sterner 2001), as illustrated in Figure 20-3. In most cases, TCE substitution is relatively cheap, but it may be expensive under special circumstances: if an individual firm lacks space (water-based equipment is typically larger), time, or finances (i.e., liquidity problems or difficulty borrowing money for this kind of investment). In addition, in small companies, all personnel may be “essential,” and even one week lost on “side issues” such as degreasing can carry a significant cost that is hard to quantify but may be reflected in strong feelings about the ban. Requiring firms to comply quickly may also exacerbate costs considerably (see Chapter 9).

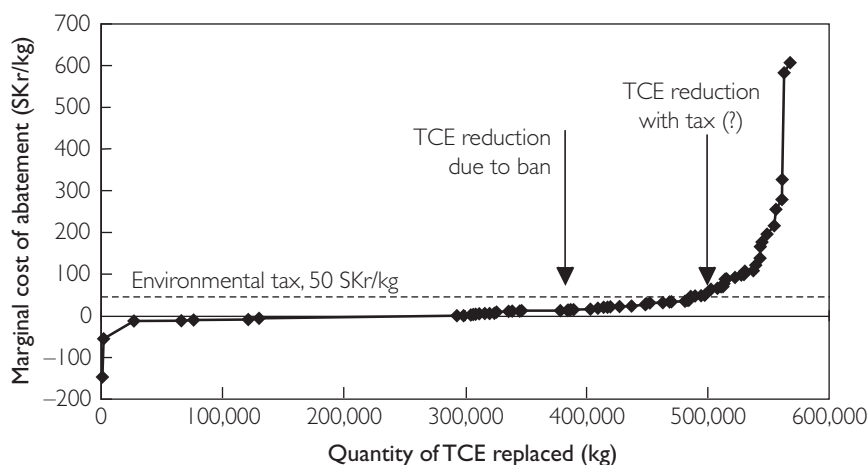


FIGURE 20-3 Marginal Abatement Cost and Effects of Tax Compared with Ban on TCE in Sweden

Note: SKr = Swedish krona (1 SKr ≈ US\$0.10). Source: Data from Slunge and Sterner 2001.

It is difficult to distinguish the firms that are bluffing from those that really face compliance difficulties. Companies that face tough environmental restrictions often protest that the requirements are too tough, but they sometimes find it easier than expected to make the change—and may even find that the new processes are cheaper (see Chapter 14 on the Porter hypothesis).

Is a ban a good policy instrument in this case? A traditional “P-vs.-Q reasoning” suggests that if the marginal cost of abatement is steep close to the zero limit (and the damage curve not so steep), then taxes (or other price-type instruments) would be better (Weitzman 1974). If technological progress is expected to be fast but unevenly distributed between specific application areas, then this preference for taxes would be strengthened by considerations of dynamic efficiency (see Chapters 9 and 10). When regulation is the instrument used, the company has a stronger incentive to overestimate abatement costs than when taxes are used (see Chapter 10).

Figure 20-3 shows that a large share of emissions have a low marginal cost of abatement. Some of the high estimates may be tactically exaggerated, but at least 90% of the abatement probably would have been achieved with an environmental charge of 50 SKr (about US\$5 with 2001 exchange rates) per kilogram. The reduction with the ban has yet to reach 90%. An increase to a higher charge would not have had much of an effect because the costs of abatement rise dramatically at the end.

Norway

Since 2000, Norway has taxed TCE and PER at a rate of 50 Norwegian crowns or US\$6/kg. Those taxes raised the market price about fivefold and may be assumed to have a large effect, as suggested by firm-level inquiries (Slunge and Sterner 2001). Industry has not been enthusiastic but is still fairly complacent, apparently appreciating that the tax allows firms much more flexibility than a ban. Data for 2000 indicated a drastic decline in TCE use of more than 80%. Even if there was some increase in 1999 (attributed to pre-tax hoarding), the fee still appears to have been effective.

The fee has partial deposit–refund characteristics in that half of it is refunded on delivery of TCE-contaminated sludge for proper treatment. This feature reduces net payments from the industry to the state, which reduces political resistance. It also specifically targets emissions rather than use, making closed systems attractive. Because the emissions cause health and environmental problems, this approach seems logical. The fee proceeds also could have been used to support research into alternative degreasing methods or even to provide loans on favorable terms for special cases. This alternative runs counter to the usual public economics arguments against earmarking, but these funds are both minor and transient. The interest for the treasury is bound to be minute, and the politically important aspect is to gain the support and understanding of—and preferably, partnership

with—the industry concerned to avoid the kind of confrontation provoked by the Swedish ban.

Germany versus Sweden and Norway

The Swedish TCE ban has not been fully effective. The legal battles were costly, and SEPA lost prestige in the eyes of the complying companies that followed regulations in good faith only to find their efforts “unnecessary” when their competitors did not even have to pay environmental or compensatory fees. Prohibition should only be used in clear-cut cases (when health or environmental damage is specifically targeted or international opinion more coordinated).

Other policy alternatives worth considering are stricter emissions requirements, a deposit–refund scheme, and special environmental charges distinct from taxes. Germany has neither tax nor prohibition but tough technological requirements concerning emissions instead. This approach has led to a much more rapid decrease in TCE use than the European average—even faster than in Sweden (see Figure 20-4). It has also led Germany to develop new technology for a very successful line of degreasing machines that recycle all solvents and that are now the foundation for more German exports in this area (a nice illustration of Porter’s ideas).

SEPA probably would have been more successful with a fee, as in Norway, but perhaps applied over a somewhat broader range of chemicals. If the proceeds had been earmarked for environmental collaboration with the industry association, then the results might have been even more enhanced.

The conclusion must also be seen in the broader context of all chemical hazards. Legal and administrative systems could never handle such complicated, expensive, and time-consuming conflicts concerning each individual chemical. Many thousands of industrial chemicals are as dangerous as (or more dangerous than) TCE. Traditionally, the regulation of chemicals has suffered from new source bias because the new chemicals have to pass

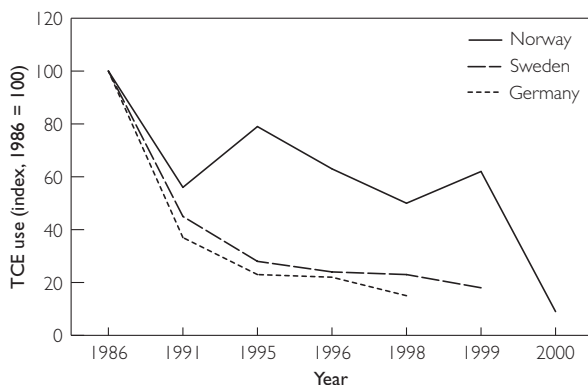


FIGURE 20-4 Use of TCE in Norway, Sweden, and Germany

complicated tests and administrative barriers to be registered, whereas older chemicals continue to be used by default. (Gasoline is one example; some of its ingredients, such as benzene, are extremely toxic and carcinogenic.)

In a current review of its chemical strategy, the European Union has started to discuss reform in the way chemicals are tested, which would mean that chemicals would be categorized by their level of hazard. For those that fall in the more dangerous (carcinogenic, mutagenic, or bioaccumulating) categories, the requirements for testing will be made much more stringent, and the responsibility will be placed squarely on the producers and importers.

Global Policymaking: Protecting the Ozone Layer

International response to scientific evidence that the stratospheric ozone layer is being depleted by the emissions of anthropogenic ODSs has been fairly rapid. It is a good illustration of the process of creating rights to the environment by international agreement (described in Chapters 3 and 8). First, in 1985, several nations agreed to form the Vienna Convention for the Protection of the Ozone Layer. The Montreal Protocol on Substances that Deplete the Ozone Layer, which builds on the Vienna Convention, was agreed upon later that year and proved to be far tougher than many observers would have expected only a few months earlier. It came into force, on time, on Jan. 1, 1989. By then, 29 countries and the European Economic Community—representing approximately 82% of world consumption—had ratified it. More than 170 countries (including about 130 developing nations) are now parties to the convention and the protocol.

The negotiations were diplomatically tricky because the distribution of gains and costs is uneven despite the global nature of the problem. Stratospheric ozone (as opposed to tropospheric or ground-level ozone) plays an important positive role by absorbing most of the biologically damaging ultraviolet sunlight (UV-B), stopping it from reaching the Earth's surface. Many experimental studies of plants and animals have shown the harmful effects to humans of excessive exposure to ultraviolet radiation. Ozone depletion in the stratosphere leads to more ultraviolet radiation.¹¹ The effect is unevenly distributed: hardly any effect at the equator and large effects toward the poles (areas in which mainly high-income people live). The effect of moderate radiation will primarily be an increase in skin cancer for people with fair skin—again, a medical problem mainly for rich, fair people who live long enough to worry about cancer. For people with darker skin in countries where large percentages of children still die of preventable illnesses such as diarrhea, basic health issues such as clean water and refrigeration (to keep food and medicines fresh) are much more important. If the benefits are unevenly distributed to the advantage of the rich, it is unfortunately the opposite with some of the costs, because poor people are forced to pay more for refrigeration

and air conditioning. Thus, in essence, the treaty asks poor people to pay more for a benefit that would largely accrue to richer countries.

The agreement had to take into account these differences between countries. The 1990 London Amendment to the protocol contains clauses that allow developing countries certain delays and that create mechanisms for transferring technology and funds (the Global Environment Facility, or GEF) to cover the special circumstances of several groups of countries, especially developing countries. The protocol is also flexible; it allows for adjustments in case the scientific evidence changes, without having to completely renegotiate the agreement. Instead, new amendments are added, for instance, relating to new chemicals with a weaker ozone-depleting effect.

After the Montreal Protocol, most countries adopted individual plans for the phaseout of ODSs. In the major producing countries such as the United States, the main policy was a fairly brisk progressive prohibition of both production and consumption of ODSs. According to Section 604 of Title VI in the U.S. Clean Air Act, “it shall be unlawful for any person to produce any class I substance in an annual quantity greater than the relevant percentage specified....” The text goes on to give specific percentages for each year and chemical (e.g., for carbon tetrachloride, 100% in 1991, 90% in 1992, 80% in 1993, 70% in 1994, 15% in 1995–1999, and 0% thereafter). The most prominent ozone-depleting chlorinated solvents were regulated in turn, depending on their ozone-depleting potential and areas of use. To create some flexibility and facilitate the phaseout, permits could be traded but not banked. Because permit allocation was based on grandfathering and provided an advantage to some firms, policymakers imposed taxes as a complement to limit windfall profits (see Tietenberg 1995). These taxes were fairly important and collected almost US\$3 billion in the first five years. They appear to have been an important factor in decreasing the use of ODSs.¹²

The main policy instruments used in ODS-importing countries such as Sweden were import restrictions. Between 1988 and 1994, the use of ODSs in Sweden decreased by 93%; thus, the Swedish strategy was fairly successful—also fairly easy, because there was no domestic production.

Global Climate Change: International and Domestic Policies and Carbon Markets

The largest international environmental issue at present is global climate change. Because the effects of the targeted pollutants are truly global, policymaking must be international, so the structure of international negotiations becomes important. (The theory behind these negotiations is described in Chapter 14.)

In practice, the history of international climate negotiations is long and complex. In 1992, the United Nations Framework Convention on Climate Change (UNFCCC) established broad goals for climate protection and

sustainable development and created the milieu for international negotiation on more specific goals and measures. In December 1997, the parties to the UNFCCC agreed on the terms of what came to be known as the Kyoto Protocol. This agreement established emissions commitments for industrialized countries for the 2008–2012 time frame, the so-called first commitment period. However, it did not settle all climate issues; negotiations at the next four Conferences of the Parties (COPs) addressed a variety of implementation details. After the 2001 COP in Marrakesh, Morocco, industrialized countries began to ratify the Kyoto Protocol. Nevertheless, by that time the George W. Bush administration in the United States had declared that it would not ratify the Kyoto Protocol. The government of Australia soon thereafter echoed its lack of support. This illustrates how difficult it has been for the world community to take steps toward reducing greenhouse gas emissions. The reasons for the difficulty are that climate change is uncertain and complicated (ecologically and technologically), abatement (and damages) will be costly, and the distribution of the cost burden is a contentious issue. Nevertheless, despite the withdrawal of these two countries, the Kyoto Protocol entered into force in 2005, having met the dual requirements that 55 countries had ratified the agreement.

A grand bargain to secure acceptance by countries with concerns about the economic burdens of these targets included an array of market-based mechanisms to promote cost-effectiveness implementation (Aldy and Stavins 2007). Through the Kyoto Protocol, three “flexible mechanisms” were designed to enable CO₂ emission reductions to occur in the cheapest locations across the world. The first mechanism, emissions trading, can occur between countries with binding targets; that is, the so-called Annex 1 countries (mainly industrialized countries, members of the OECD, plus the transitional economies of central and eastern Europe and the former Soviet Union) that agreed to various percentage reductions (averaging around 5%) in national emissions relative to a 1990 baseline by 2008–2012. These countries are allowed to meet their domestic targets by purchasing credits from other countries that have exceeded their targets in the EU emissions trading system (ETS), the largest carbon market implemented so far (see Chapter 6).

Mechanism Two is the Clean Development Mechanism (CDM) that allows credits from emission reduction projects in poorer countries to be used in rich countries to partially meet their own commitments under the Kyoto Protocol. Certified Emissions Reductions (CERs) from CDM are project credits flowing into the global compliance in the EU ETS market generated through emission reductions. It is possible to buy CERs directly from projects that reduce CO₂ emissions—then called “Primary CERs”—or from companies who have bought Primary CERs and are selling part of their portfolio to third parties—then called “Secondary CERs.”

Third is the Joint Implementation mechanism (JI), also a project-based mechanism that allows countries with binding targets to get credits from projects carried out in other countries with binding targets. JI differs from the

emission trading mechanism in the EU ETS in the sense that more industry sectors are included. Besides, JI allows credit for reducing any of the six major GHGs, whereas the EU ETS only covers carbon dioxide.

In addition to the mechanisms under the umbrella of the Kyoto Protocol, a number of state-based and regional initiatives have emerged over the past few years with the goals of implementing emissions trading programs and influencing any federal or national scheme (Brohé et al. 2009). For example, Australia's New South Wales (NSW) has in operation a program called the NSW Greenhouse Gas Abatement Scheme (GGAS) to reduce GHG emissions until 2012 from the power sector. Retailers and large electricity consumers in NSW and in the Australian Capital Territory are required to meet mandatory intensity targets to reduce or offset the emissions of GHG arising from the production of electricity they supply or use. They can meet their targets by purchasing credits that are generated through the following activities: low-emission generation of electricity and improved generation efficiency, activities that result in reduced consumption of electricity, or on-site generation of electricity and carbon sequestration. In June 2007, Prime Minister John Howard announced that the Australian government would introduce a domestic emissions trading scheme (the Australian Carbon Pollution Reduction Scheme, CPRS) by 2012 at the latest. The cap of the CPRS would be consistent with a long-term GHG reduction goal of 60% or more below 2000 levels by 2050 and a mid-term target corresponding to a reduction in emissions of 5% to 25% below 2000 levels by 2020, with the more stringent target being conditional on an international climate agreement including similar targets for other developed countries, as well as substantial emission restraint in major developing economies. All emissions of Kyoto GHGs from industrial processes, stationary energy, transport, waste, and fugitive emissions would be covered from the start, while agriculture could be phased in in 2015. Reforestation activities could be opted-in on a voluntary basis while emissions from (domestic) deforestation will most likely be addressed through other incentives (for the inclusion of offsets that reduce emission from deforestation in other countries, see Chapter 27). Nevertheless, the government recently decided to delay plans until the Kyoto Protocol expires in 2012.

Under the Regional Greenhouse Gas Initiative (RGGI), ten northeastern and mid-Atlantic states in the U.S.—Connecticut, Delaware, Maine, Maryland, Massachusetts, New Hampshire, New Jersey, New York, Rhode Island, and Vermont—committed to reduce power sector CO₂ emissions by 10% below 2009 levels in 2019. RGGI is composed of individual CO₂ budget trading programs in each of the ten participating states. These ten programs are implemented through state regulations and linked through CO₂ allowance reciprocity. Regulated power plants can use a CO₂ allowance issued by any of the ten participating states to demonstrate compliance with the state program governing their facility. Taken together, the ten individual state programs function as a single regional compliance market for carbon emissions. Forward

progress in the U.S. has been slow and uneven; Pennsylvania is considered actively joining RGGI, while the governor of New Jersey announced his state would leave RGGI by the end of 2011.

Some other initiatives have been under analysis or early development. For instance, the Western Climate Initiative (WCI) covers a group of seven U.S. states and four Canadian provinces, with an aggregate emissions target of 15% below 2005 levels by 2020. The Midwestern Greenhouse Gas Reduction Accord (MGA) covers nine Midwestern governors and two Canadian provinces that have agreed to establish regional GHG reduction targets, including a long-term target of 60%–80% below current emission levels and develop a multisector cap-and-trade system to help achieve the targets.

On November 2009, New Zealand's parliament passed the Climate Change Response Amendment, establishing a New Zealand ETS program. During the transition period (2010–2012), there will be no aggregate cap on emissions. Entities will be accountable for their emissions, however, and will need to purchase allowances (at a fixed price) when emissions exceed their free allocation. As this book went to press, the New Zealand ETS was undergoing an independent review that would take into consideration the overall effectiveness of the scheme, new schemes in place in other developed countries, and the potential for linking to these schemes (World Bank 2010).

In October 2008 Japan launched a trial domestic scheme based on voluntary participation. In addition, in 2010, it proposed the “Basic Act on Global Warming Countermeasures.” This policy framework is in line with Japan's mid- to long-term GHG emissions reduction targets. The Act considers establishing a mandatory ETS, introducing a carbon tax, and putting in place a feed-in tariff for all renewable energy sources. The ETS would start within one year after the Act comes into force (World Bank 2010).

Finally, voluntary markets for emissions reductions that are not compliant with the Kyoto Protocol are also available for sale to corporations and individuals who want to offset their emissions for nonregulatory purposes, especially to cover emissions from air travel. Emission offsets in this latter category are verified by independent agents, but they are not certified by a regulatory authority for use as a compliance instrument. Commonly referred to as Verified Emission Reductions (VERs), they are not a standardized commodity; many of these reductions have no secondary market benefits outside of their embedded “green image value” or speculative value. Although some commentators forecast that voluntary markets could eventually overtake compliance markets, it seems unlikely that altruistic markets alone could form the main driver of global emission reductions.

The Chicago Climate Exchange (CCX) belongs also to the category of voluntary initiatives. As described in Chapter 6, the CCX had nearly 300 members—mainly multinational and multisector companies as Motorola Inc., Ford Motor Co., IBM, Intel Corporation and DuPont—that made a voluntary but legally binding commitment to reduce GHG emissions. While the EU ETS,

the CDM, and JI are integrated markets, no other fledgling market is currently linked with the EU ETS. However, some possibility exists that other carbon markets will be linked in the future. Indeed, RGGI provides a safety valve on the upward pressure of prices. If prices exceed an average of US\$10/ton in a 12-month period, firms may purchase international offset credits in the form of ERUs—emission reduction units—or CERs. This safety valve may offer the first opportunity to link the EU ETS and CDM with a U.S. carbon market.

There are major advantages to linked carbon markets. A linked market might establish related carbon prices in different regions and reduce any competitive distortion arising from different pre-link carbon markets among partners while entailing a signaling of a commitment to further cooperation on climate change policy. In the absence of administrative restrictions, direct multilateral linking should lead to full price harmonization across the linked schemes. Furthermore, deepening the carbon markets is likely to reduce transactions costs, increase market liquidity, and increase efficiency and cost savings simply by redistributing emission reductions across linked systems without affecting the aggregate level of remaining emissions under those systems. However, a linked system might import volatility from partner systems (McKibbin et al. 2008) and have some distributional impacts since allowance trading resulting from linking will raise the allowance price in one of the linked schemes while reducing the other scheme's allowance price. The final impacts on each participant will depend on whether the participant is a net buyer or seller of allowances and on the extent the price changes. Since prices of energy and other emissions-intensive goods are also linked to carbon prices, they will rise in the case of the participants whose allowance price rises, affecting their competitiveness, and they will fall in the case of those participants whose allowance price falls (McKibbin et al. 2008). We might add that while linking is attractive, the gains could be illusory. If negotiators could not solve a number of issues such as allocation and burden sharing necessary to create a large trading area, then it is not unlikely that these issues will turn up (in some new guise) if the regulators first create separate schemes and then "link" them. For instance the price and trade movements in response to different levels of stringency could well lead to political critique.

Review of the state and trends of carbon markets—compressing both mandatory and voluntary initiatives—reveals some interesting facts (Capoor and Ambrosi 2006; 2007; 2008; 2009).

- The overall carbon market has grown steadily since 2005, reaching a total value transacted of about US\$126 billion at the end of 2008 (see Table 20-5). The vast majority of this volume has been exchanged through the EU ETS scheme, followed by CDM. Although CERs are effectively fungible with emissions reductions in the EU ETS, they have been traded at a variety of discounted prices as a result of the various risks that may be applicable. Such risks include the project delivering fewer emission

TABLE 20-5 State and Trends of Carbon Markets

	2005				2006				2007				2008			
	Volume (MtCO ₂ e)	Value (MU\$)	Price (US\$/ tCO ₂ e)	Volume (MtCO ₂ e)	Value (MU\$)	Price (US\$/ tCO ₂ e)	Volume (MtCO ₂ e)	Value (MU\$)	Price (US\$/ tCO ₂ e)	Volume (MtCO ₂ e)	Value (MU\$)	Price (US\$/ tCO ₂ e)	Volume (MtCO ₂ e)	Value (MU\$)	Price (US\$/ tCO ₂ e)	
Allowance Markets																
EU ETS	321	7908	24.64	1101	24357	22.12	2060	49065	23.82	3093	91910	29.72				
New South Wales	6	59	9.83	20	225	11.25	25	224	8.96	31	183	5.90				
Chicago Climate Exchange	1	3	3.00	10	38	3.80	23	72	3.13	69	309	4.48				
RGGI	na	na	na	na	na	na	na	na	na	65	246	3.78				
Sub-Total	328	7970	24.30	1131	24620	21.77	2108	49361	23.42	3258	92648	28.44				
Project-Based Markets																
Primary CDM	341	2417	7.09	450	4813	10.70	552	7433	13.47	389	6519	16.76				
Secondary CDM	10	221	22.10	25	444	17.76	240	5451	22.71	1072	26277	24.51				
Joint Implementation	11	68	6.18	16	141	8.81	41	499	12.17	20	294	14.70				
Voluntary Market	20	187	9.35	17	79	4.65	43	263	6.12	54	397	7.35				
Sub-Total	382	2893	7.57	508	5477	10.78	876	13646	15.58	1535	33487	21.82				
Total	710	10863	15.30	1639	30097	18.36	2984	63007	21.11	4793	126135	26.32				

Source: Capoor and Ambrosi 2006, 2007, 2008, and 2009; State and Trends of The Carbon Market, The World Bank.

reductions than planned, or problems or delays arising in the UNFCCC administrative process. For the same reason, since secondary CERs come without the risks of primary CERs, they are traded at higher prices.

- CDM accounts for most of the project-based market activity, with JI and the voluntary market having just a small role. Nevertheless, primary CER transactions have been constrained lately by the financial crisis and questions about the rules of eligibility post-2012, declining nearly 30% in 2008 (from nearly 552 million CERs in 2007). Many of the project-based addressing post-2012 credits are still in the proposal stage and likely to be influenced by the outcome of forthcoming international climate negotiations. The JI market has also experienced a slowdown, ending 2008 with just half the volumes transacted in 2007.
- Asia is the largest supplier of CERs, followed by Latin America, developed economies, and Eastern Europe. For example, in 2008, China accounted for 84% of the transactions in the primary CDM market, followed by India and Brazil with 4% and 3% market share, respectively. At the end of the spectrum, a number of countries in sub-Saharan Africa have started to supply CERs in spite of the infrastructure bottlenecks that constitute a major constraint for development. In the case of JI, Eastern European countries concentrate most of the projects; Russia accounts for 68% of the transacted volume, followed by Ukraine with an 18% market share.
- When it comes to the sources of CERs and JI, projects in hydro, wind, biomass, and energy efficiency of power generation at large industrial facilities lead the growth of CDM, accounting for more than 70% of the total number of projects in 2008.

In many respects, the CDM has been the success story of carbon trading to date because it created incentives to mitigate GHG emissions globally through international carbon finance transfers. However, in spite of CDM's achievements in reducing emissions in developing countries, it has come under attack for various reasons (see Michaelowa and Jotzo 2005; Müller 2007; Hepburn and Stern 2008). First is the issue of additionality and the risk that CDM might create some perverse incentives: the Kyoto Protocol specifies that emission reductions are only to be certified under CDM if they are additional to any that would occur in the absence of the certified project activity. The CDM requires additionality to assure that CO₂ emitted in developed countries against a CER is not increasing the total level of emissions permitted under the trading scheme. However, since there is no baseline against which such additionality can be assessed, governments might have an incentive to not impose regulations on emissions if this means that environmental projects become lucrative without the support of carbon finance through the CDM.

Second, transaction costs of the CDM have been rather high because of the requirements to demonstrate project additionality on a case-by-case basis. In practice, additionality is a complex concept since it involves speculation about

what would have happened in the absence of the project obtaining carbon finance. Finally, CDM can only serve as transitional mechanism because it does not generate emission reductions above and beyond those required by targets for developed countries. Although such problems are no doubt serious, CDM has nevertheless provided a very important basis on which to construct a more sensible and effective global climate policy. Maybe its biggest contribution to date has been to capture the interest of governments and companies in developing countries in viewing climate change mitigation as an opportunity instead of a constraint to growth.

Another important contribution has been to provide experience and lessons on what has worked well and what could improve if market mechanisms are to be relied on to provide further mitigation. Finally, it seems that CDM has contributed to technology transfer. Seres et al. (2009) and De Coninck et al. (2007) provide comprehensive analyses of technology transfer in the CDM to date. They point out that CDM has encouraged technology transfer in practice; technology transfer is more common for larger projects and projects with foreign participants; and it is very heterogeneous across project types, usually involving both knowledge and equipment. In general, hardware technology transfers take place to a larger extent in projects that reduce non-CO₂ GHGs than in renewable energy and energy efficiency projects. The exception seems to be wind energy, where all projects use technology from the EU.

What is to Come in the Post-Kyoto World?

The Kyoto Protocol was intended to be a first step through 2012 to mitigate the threats posed by climate change. However, a second step is needed, and policy-makers, scholars, businesspeople, and environmentalists have begun debating the structure of the successor to the Kyoto Protocol. The challenge for the international policy community lies in deciding on the next step, in terms of both climate-related goals and the design of policies to implement those goals. Setting climate change policy goals presents perhaps the greatest challenges to the international policy community. The emission commitments for the Kyoto Protocol's first period were among the most contentious elements negotiated at the Kyoto conference. Calls for more ambitious targets, broader participation in the number of countries with obligations, and expansion of the sectoral coverage of carbon trading—particularly with regard to aviation—have grown louder. Certainly, the most glaring weakness in the Kyoto Protocol is obvious: Three of the five largest GHG emitters in the world do not face constraints in their emissions. China and India do not have quantitative emission targets, and Russia's Kyoto Protocol commitment is so generous that it is unlikely to bind before 2012. In addition, the largest GHG contributor, the United States, did not ratify the agreement. These omissions raise questions about the future of climate negotiations and the incentives for participation

in the post-Kyoto agreement. On the one hand, without the participation of major developing countries—who are likely to become primary industrial emitters in the future—emissions abatement by industrialized countries will not do much to mitigate climate change: leakage might induce a relocation of carbon-intensive industries from countries with emissions commitments to nonparticipating countries, or might cause increased consumption of fossil fuels by nonparticipating countries in response to declines in world oil and coal prices. On the other hand, developing countries continue to press the claim that the Annex I countries should do even more to reduce GHG emissions given their historical responsibility.

Despite existing concerns that climate change negotiations are already excessively complicated, it may be possible to promote more participation by enlarging the issues under consideration (see Chapter 14). Integrating climate change measures in the development agenda and the trade agenda may serve as opportunities for effective issue linkage. The Montreal Protocol allowed for trade sanctions; climate change policies could also incorporate such penalties. Some analysts think we should break up the gigantic climate negotiation into separate treatments for forestry, technology, adaptation, and maybe for different gases (Barrett 2008).

While the debate is ongoing, climate change is occurring, and it will continue to occur to some extent regardless of GHG emissions mitigation efforts. The IPCC (2007) forecasts accelerated warming under a variety of scenarios. Even if atmospheric concentrations of GHG could be held constant at year 2000 levels through the 21st century, the global climate would still warm 0.6°C ($+/-0.3^{\circ}\text{C}$). Under a variety of long-term emission scenarios, temperature increases could even range from 1.1°C to 6.4°C by 2100. Therefore, the new climate policy architecture should support adaptation efforts identifying the needs for those countries that lack the capacity and resources to adapt adequately to the changing climate. The architecture should also focus on promoting R&D efforts on emission mitigation and adaptation technologies. Stabilizing long-term atmospheric GHG concentrations will require the development and deployment of low-carbon and eventually zero-carbon-emitting technologies on an extensive scale. Investments in adaptation R&D can deliver an array of benefits in mitigating the risks of climate change. Some of these efforts could focus on protecting developed coastal areas against sea-level rise and storm surges under climate change. R&D focused on the human health impacts of climate change—for example, a malaria vaccine—could reduce the risks of climate change and improve public health in low-income countries in a way that could facilitate economic development. Finally, the post-Kyoto policy architecture should be flexible enough to adapt to new information about climate science, as well as technological factors. It should allow for informed updating of objectives in response to new information.

Current proposals for the EU attempt to strengthen and expand the EU ETS beyond 2020; this is the core element of the Climate and Energy

Package adopted by the European Parliament in December 2008. Its main objectives are: to reduce overall GHG emissions to 20% below 1990 levels by 2020 (possibly scaling up to 30% in the event of satisfactory international response); to increase the share of renewable energy sources to 20% by 2020; and, to improve energy efficiency 20% by 2020.

The Kyoto Protocol was the first step to address climate change; as such it was not perfect. It is likely that the next step is not going to be perfect either. However, it might be a step toward a long-term improvement, providing the basis for continued efforts to address the problem and securing sufficient international political support to move forward.

Supplemental Reading

Policy Instruments

Aldy and Stavins 2007
 Carlson et al. 2000
 Cason 1995
 Kolstad 2000a
 Metcalf and Weisbach 2010
 Michaelowa 2004

Climate Change Risks and Policies

Anger 2008
 Bassett and Frankel 2010

Cao 2010
 Dietz and Maddison 2009
 Helm 2008
 Lutsey and Sperling 2008
 Olmstead and Stavins
 2010
 Schmidt et al. 2008
 Toman 2001
 Ulph and Ulph 2007
 Zenghelis and Stern 2009

Notes

- 1 To give room for new material, a number of sections from the first edition are omitted from this edition but available at the book's website, www.economics.gu.se/policyinstruments. This applies to environmental tax reform in Sweden, Toxic Release Inventory and other voluntary programs, and also the Superfund and liability for toxic waste disposal.
- 2 This subsection and the next benefited from helpful comments from Dallas Burtraw of Resources for the Future.
- 3 Much of the reduction in sulfur use in Sweden predates the tax. Emissions were more than 900 kilotons in 1970, 500 kilotons in 1980, and 136 kilotons in 1990. Only the last (but perhaps most difficult) reduction, to 66 kilotons in 1999, was (partly) attributable to the tax. The Swedish tax applies to sulfur emitted but is levied on fuels on the basis of their sulfur content; a rebate is provided for sulfur removed by filters and so forth.
- 4 The emissions rights were called "allowances" but sometimes are called "permits" or "rights." They were not property rights in the formal sense because, in principle, the U.S. Congress could reduce or even abolish them without the owners being able to raise compensatory claims (Joskow and Schmalensee 1998).

- 5 The figures apply to Phase 2. The initial phase (Phase 1, 1995–1999) was a transition that focused on reducing emissions at a few large firms. The program also includes separate targets of 2 million tons of NO_x reduction.
- 6 This section is based mainly on Höglund 2000, Sterner and Höglund 2000, and SEPA 1997.
- 7 The sulfur oxides tax and the nitrogen oxides charge are of similar magnitude. The sulfur tax is US\$3/kg of sulfur (the same as US\$1.50/kg of SO_2), whereas the NO_x tax is US\$4/kg of NO_2 (US\$2.60/kg of NO). The two most common oxides, SO_2 and NO, happen to have roughly the same acidifying effect per unit of weight (1,000 grams of NO gives $1,000/30 \approx 33$ moles each of NO and H^+) because the molecular mass of NO is 30. Similarly, SO_2 gives $1,000/64 \approx 16$ moles of SO_2 , but $1,000/32 \approx 31$ moles (twice as many) of H^+ because there are two hydrogen molecules in sulfuric acid. However, HNO_3 may be denitrified in nitrogen-deficient ecosystems, reducing its acidifying effect. On the other hand, NO_x also leads to eutrophication and the formation of ground-level ozone (the main concern in California, but not in Sweden).
- 8 Many countries use other instruments, such as standards based on (implicit) shadow tax rates. The Dutch ministry informally recommends a value of US\$4–\$5/ton to local authorities for rule setting. If abatement technologies are less expensive, then the local government should insist on their adoption. The Dutch government has discussed a *Kosten Verevening* (settlement) program in which the higher-than-average polluters would pay those below average, which is akin to a refunded emissions program or a permit program (Kroon 2000).
- 9 This section is based on Slunge and Sterner 2001.
- 10 The fact that regulations and laws governing the use of TCE were not coordinated across EU countries was an important factor in favor of allowing Sweden's regulations. In areas of law where there is explicit harmonization (e.g., pesticides or vehicles), it would be harder to accept different national legislation.
- 11 Over Antarctica, up to 60% of the total stratospheric ozone is depleted during the spring (September to November). Scientific evidence shows that synthetic chemicals are responsible for the observed depletion. The ODSs are hydrocarbons combined with one or more halogen atoms (e.g., chlorine, fluorine, or bromine) and often are called *halocarbons*. One important group are the chlorofluorocarbons (CFCs). CFCs were first produced in 1928 and became popular because they were usually less hazardous than the chemicals they replaced, partly due to their stability—the very attribute that lets them survive until they manage to reach the stratosphere. CFCs include carbon tetrachloride and methyl chloroform, which were used in refrigeration, air conditioning, foam blowing, and cleaning of electronic components and as solvents. Today, CFCs have been replaced by HCFCs (hydrochlorofluorocarbons), which are only partly halogenated and thus less hazardous to the ozone layer (although not harmless, either). Another important group is the halons, used in fire extinguishers.
- 12 The United States started to tax selected ODSs in 1990. The tax was determined by multiplying a base rate per pound of ODS by an ozone-depletion factor. Initially set at US\$1.37/lb, the base tax gradually increased to US\$5.35/lb in 1995 and then by US\$0.45/lb per year. Ozone-depletion factors based on each chemical's relative damage were originally set in the Montreal Protocol. For example, methyl chloroform has a factor of 0.1, whereas halon-1301 has a factor of 10.0, so in 1995, the tax on methyl chloroform was US\$0.53/lb and on halon-1301 US\$53.50/lb.

Experience in Developing Countries

In developing countries, the first priority in policymaking is to provide employment and income opportunities. Pollution is typically thought to grow as income grows—at least at the lower end of the Kuznets curve (see Chapter 2). In many developing countries as well as in the more industrialized of the formerly planned economies in central and eastern Europe (CEE), pollution from industries creates the majority of emissions.¹ In the richer countries, environmental protection agencies typically targeted industrial pollution first because industries are relatively easier to control (because they are large, immobile, visible, and have at least some economic resources at their disposal). Most of the environmental protection agencies of poorer countries do likewise and start by regulating industries. One difference is that these agencies typically are more aware of the clean technologies that are available than the OECD countries were when they started their process of cleanup.²

It is sometimes asserted that developing countries are not interested in pollution control (e.g., World Bank 2000), but this assertion is little more than prejudice. In fact, many developing countries put considerable emphasis on pollution control, and although the stringency of control increases directly with rising income, this process starts at low income levels (Dasgupta and Mäler 1995). Most of the decline in pollution intensity comes at the low rather than the middle income level (Hettige et al. 1996). However, experience also shows considerable individual variation from one country to another depending on culture, politics, climate, and many other factors.

In this chapter, we illustrate the range of opportunities that exist by looking at some successful environmental policy instruments. Variations in instrument design are closely tied to variation in economic policy in general, and politics differ widely in developing countries—from a strong dedication to market principles to belief in import substitution and infant-industry protection, not to speak of the remaining partly planned economies. Many countries continue to subsidize energy and protect domestic production of polluting goods.

Given the diversity of policymaking, it is difficult to generalize about the experiences of developing countries, but one fact does appear to apply broadly: physical regulation may be a starting point for environmental policymaking,

but it is commonly insufficient where institutions are weak. Regulation is complicated, because emissions vary and are difficult to measure (even for point sources). If the environmental protection agency has little staff and other resources, the monitoring equipment is prohibitively expensive, and staff are underqualified and underpaid, then the risks of deceptive lobbying or even corruption are particularly high. Given the urge to industrialize and to attract investments, policies that build on tough enforcement are not attractive.

Experience shows that abatement costs in developing countries typically are low. Sometimes they are lower than managers think—especially if firms are given some suitable advice and sufficient time for flexible adaptation; mild economic or informational policies can be effective. In a situation where drastic fees or prohibitions would lead to antagonistic or deceptive responses and might be fought in court, small fees coupled with assistance and information may lead to a process in which firms voluntarily engage in substantial abatement efforts. Instruments that build mainly on reputation, such as labeling schemes, appear to have similar effects. In many cases, the best policy is a combination of instruments. This chapter addresses several topics:

- environmental fees and funds in China;
- environmental charges in Rio Negro, Colombia;
- tradable permits in Chile;
- voluntary agreements in emissions control by Mexican firms;
- differentiated electricity tariffs in Mexico and Zambia;
- information provision in Indonesia (PROPER); and
- zoning and self-regulation of industrial estates in India.

The first two sections are dedicated to the environmental charges and funds. These schemes are not simple Pigovian taxes; rather, they are combined with other instruments, and much attention is placed on the use of the tax proceeds. The third section focuses on tradable permits to control pollution coming from point sources. The fourth section focuses on voluntary agreements and other policies in the informal manufacturing industry. The fifth section focuses on the design of tariffs in developing countries. The two last examples are intended to emphasize the importance of building appropriate institutions and of choosing policy instruments that are adapted to the resources at the disposal of a local environmental protection agency.

Environmental Fees and Funds: China

China is not only the most populous country on the planet but is also still a fairly low income nation and the most polluted³. In this kind of a setting, economic allocation choices really become harsh. Although people would benefit from environmental cleanup, scarce funds also could be used to provide health care

and education, improve housing, and ensure a clean water supply. Atmospheric pollution is so bad that in many urban areas, atmospheric concentrations of pollutants such as particulates and sulfur dioxide routinely exceed World Health Organization (WHO) safety standards by very large margins. Thus, while the WHO standard for particulates is below $100 \mu\text{g}/\text{m}^3$, average concentrations for a sample of 50 Chinese cities are about $300 \mu\text{g}/\text{m}^3$ (Wang and Wheeler 2000).

Chinese industry is a primary source of this problem; estimates indicate that industrial pollution accounts for over 72% of SO_2 national emissions and 75% of the flue dust (Wang and Wheeler 2000). One of China's responses to this problem has been its pollution levy system, applied to all the discharges of pollutants exceeding the limits set by the state according to the quantities and concentration of the pollutants released.

China's levy system is an interesting and important example of the application of market-based policy instruments in a developing (i.e., formerly planned economy, now transitional) country. Article 18 of China's Environmental Protection Law enacted the levy in 1979. Almost immediately after, a few municipalities began to enforce the regulation, and in 1982, the state council called for a nationwide implementation. Today, most of the country is covered, and several hundreds of thousands of factories are monitored and potentially subject to this fee.

Chinese environmental fees are not textbook examples of Pigovian taxation, but few such examples exist, and many other features of the Chinese economy deviate from those in the economics textbooks, too. One distinctive feature, for instance, is that fees are paid only for discharges that exceed a certain level. The fees thus resemble a form of noncompliance fine. However, the idea of a natural resource or user fee appears to be spreading, and some of the latest water fees and the charge for SO_2 emissions are straight Pigovian charges.

Another distinctive feature of Chinese environmental fees is that they are used first to finance funds that are used by industry for abatement (roughly 70–80% of the fees) and then for central administrative control costs. After administrative costs, the rest of the funds are returned to the enterprises to be used only for environmentally beneficial uses. Thus, in a general sense, the fees also are a form of refunded emissions payment (see Chapter 7 and Chapter 20). Usually, the funds are to be used for abatement investments but may be used for current costs of environmental management. In some regions, the fund allocation is based directly on the amounts paid by the individual companies, whereas in others, it is supposedly a neutral selection of projects based on the propositions of the same firms. It is probably safe to assume that even in the latter case, firms that make large payments to the fund have some leverage in financing their own abatement projects. Thus, the firms may be less averse to paying such fees than other kinds of taxes that go only to the treasury.

Finally, the fees are relatively low and considered to be less than the marginal cost of abatement necessary to meet the Chinese emissions standards; hence, some observers question whether the fee is at all effective (NEPA 1992, 1994;

Shibli and Markandya 1995; Qu 1991). However, researchers who studied the rate of industrial emissions per unit of output for chemical oxygen demand (COD) and the effective water pollution charge for 29 Chinese provinces and urban regions between 1987 and 1993 claim that the environmental fee has had positive effects (Wang and Wheeler 2000). Supposedly, the rate is the same across the whole country—a blunt regulation that might be considered “fair”—but it is not necessarily efficient, because damage and abatement costs vary considerably across the rather heterogeneous areas of this vast country, and the pollutants are not perfectly mixed. For example, it would not be desirable to have the same standards (or charge level) in sparsely populated areas and in major cities. In practice, however, differences in monitoring and enforcement at the factory level can cause the effective fee to vary widely between provinces because some charges are only partially enforced or even waived. In this case, apparent inefficiency may improve the functioning of the overall system.

Figure 21-1 shows the significant variation in the effective levy rate among provinces. This pattern of variation is not random; effective levies are much higher in the densely populated (i.e., urbanized and industrialized) provinces of the country, particularly in the eastern coastal regions. These regions are not only more densely populated but also tend to have higher per capita incomes, which would explain why both objective damage measures and political pressures in favor of abatement would be higher. The degree of industrialization also would lead to higher pollution levels, which, under the most plausible

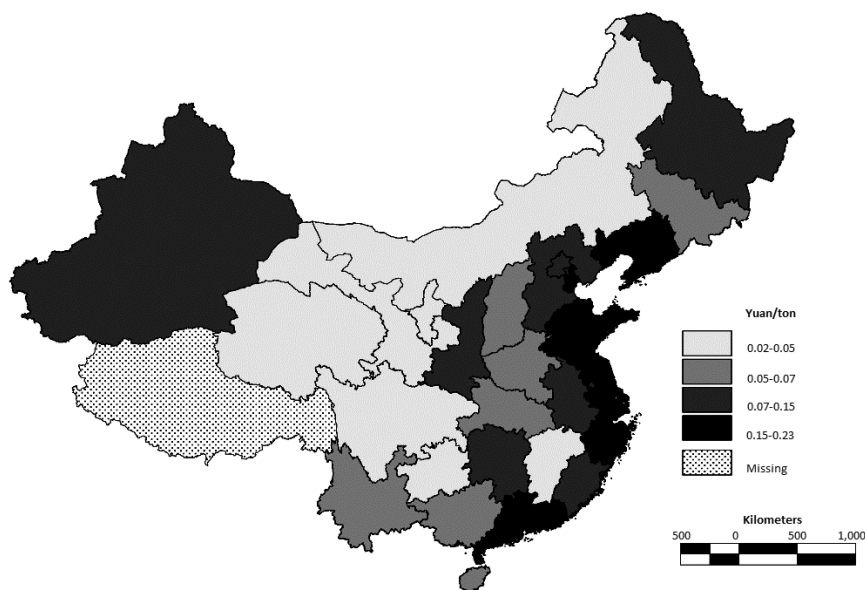


FIGURE 21-1 China's Effective Pollution Levies, 1993

Note: Levy is per unit of overstandard wastewater discharge. *Source:* World Bank 2000.

assumptions concerning the damage function, is a separate, additional factor that raises the marginal value of damages. The charge also has varied over time, with large increases since 1987. From 1987 to 1993, COD intensities fell at a median rate of 50% in all the provinces, and total COD discharges declined at a median rate of 22%.

Both supply and demand were estimated for “the environment” in a study by Wang and Wheeler (1996). The demand for “a cleaner environment” depends on population and pollution densities as well as on education, income, and share of industrial activity in economic output. The estimated elasticity of abatement with respect to the environmental fee is large and significant (almost exactly -1). Also, results indicate that larger plants pollute less and that state ownership has little independent influence on pollution intensity, which may perhaps be surprising.

Using citizen complaints as a measure, demand for environmental quality has been explained by three factors: the level of education of the affected citizens, the income level of those individuals, and the level of visible pollution (Dasgupta and Wheeler 1997)⁴. Industrial demand for the environment tends to depend on technical production intensities (e.g., wastewater intensity), production volume, and legal standards.

Management of the pollution charge may have been effective in China. Pollution levels, income, and population density appear to explain the demand for abatement. Education and bargaining power are important, too. In summary, preliminary analyses suggest that the central Chinese authorities may be wise in letting each province trade off the costs and benefits of effective implementation levels. However, despite the results presented here, many researchers doubt whether the Chinese fees are actually operative and effective. Much about the system remains to be understood.

Environmental Charges: Rio Negro, Colombia

Colombia provides an interesting example of an environmental fee in use in a less developed country⁵. The Colombian economy has been growing quickly over the past years. From 2000 to 2005, GDP grew by an average of 3.5%, and much of this growth was resource intensive (i.e., mining or agriculture). Water and air pollution also have been increasing; many of Colombia’s most important rivers are severely polluted. Most of the wastewater is untreated, and diarrhea and enteritis are the leading causes of death of children under five years old. Respiratory diseases are escalating rapidly in industrialized areas.

This section focuses on Colombia’s discharge fee system for water effluents, a program that is often mentioned as a model of well-functioning economic policy in a developing country. Law 99 passed by the National Congress in 1993 created the Ministry of Environment and local environmental protection agencies (*Corporaciones Autónomas Regionales*, henceforth

CARs) and established the legal foundation for a national discharge fee. It stipulates the use of redistributive fees to prevent, mitigate, and repair environmental damages as well as to finance the local agencies. The fees initially target effluents based only on two measures of pollution: biological oxygen demand (BOD) and total suspended solids. However, the program is more than a charge. Since it allows for heterogeneity in goal setting—to account for, among other things, differences in abatement costs and the quality of receiving waters—it also requires all key stakeholders (industries and municipalities) to actively negotiate pollution reduction targets over a five-year period. To guide the implementation of abatement measures, the Ministry of Environment set a minimum national pollution fee. Actual fees vary between regions. The regional factor starts at the national level (a factor of 1) and is increased incrementally (to 1.5, 2, 2.5 ... times the national fee) every six months until the targets are met.

Evaluations of the program have shown that lags in implementation across administrative regions have marred its performance (see Table 21-1). Some CARs initiated discharge fee programs earlier than others, and some made far more implementation progress than others (Blackman 2010). For example, in 2002, 24 out of 33 CARs invoiced fees while 21 were collecting them. Besides, collection rates varied dramatically across CARs ranging from a low of 1% to a high of 95%.

Although some lags in implementation seem natural and may even be desirable to the extent they facilitate learning by doing and the building of political momentum, the total evaluation of the fee has not been satisfactory in terms of internalization of pollution damages and changing the behavior of the polluters. Evidence suggests that the good performance was limited to the few CARs that fully implemented the program, where BOD discharges from point sources in the program fell 27% and TSS (total suspended solids) fell 45%. The reductions were much smaller in the CARs where the program was not implemented adequately, while discharges from CARs where collection of fees has not started yet continue to increase. Not surprisingly, richer and older CARs more successfully implemented the fee, which probably reflects some supply-side mechanisms: they were better equipped to supply such regulations and had more time to develop effective institutions and procedures. CORNARE, for example, recognized as one of Colombia's stronger CARs, started enforcing the discharge fee program for the Negro River in late 1997. Between 1997 and 2000, total discharges of BOD and TSS fell 62% and 90% respectively. Although these reductions may have been partly due to a series of clean production agreements signed with polluters before the program started, they are so large that it is unlikely that the program did not have a significant effect.

Noncompliance by municipal sewerage authorities was also a key barrier to successful implementation. As in many developing countries, Colombia's municipal sewerage authorities are leading sources of BOD and TSS and also leading violators of environmental regulations. The lack of compliance by this

TABLE 21-1 Participation in Discharge Fee Programs by CAR as of 2002

<i>Entity</i>	<i>No. of water users potentially covered by the fee system</i>	<i>No. of users that are invoiced</i>	<i>% potential payees that are invoiced</i>
CAM	75	45	60
CAR	491	91	19
Carder	2900	632	22
Cardique	72	54	75
Carsucre	31	10	32
CAS	91	91	100
CDA	5418	1	0
CDMB	160	153	96
Codechocó	2000	70	4
Coralina	49	6	12
Corantoquia	2607	1825	70
Cormacarena	10	10	100
Cornare	218	218	100
Corpamag	60	59	98
Corpoamazonía	22	9	41
Corpoboyacá	150	104	69
Corpocaldas	2400	610	25
Corpocesar	54	49	91
Corpochivor	170	121	71
Corpoguajira	21	21	100
Corpoguavio	23	0	0
Corpomojana	12	2	17
Corponariño	207	10	5
Corponor	49	31	63
Corporinoquía	21	0	0
Corpourabá	485	391	81
Cortolima	86	67	78
CRA	76	21	28
CRC	90	80	89
CRQ	7500	300	4
CSB	24	0	0
CVC	20000	259	1
CVS	53	16	30
Average	1383	162	48

Source: Ministry of Environment, 2002.

group undermined the “culture of compliance,” since polluters in industry and agriculture complained loudly about being made to pay the fees when many of the largest polluters failed to do so. Besides, noncompliance by municipal

sewerage authorities led to increased fees in many water basins after the 5-year total pollution load reduction targets were not met.

Some important lessons can be drawn from the experience of CORNARE. Some of the features that seem to explain its success are:

- the high level of knowledge and commitment on the part of local staff;
- solid support at higher levels, particularly the ministerial level;
- good technical information and data on pollution;
- a good working relationship with the polluting industry; and
- a relatively small region, which may facilitate monitoring.

Finally, this experience stresses the importance of compliance with environmental authorities by local authorities, which enhances credibility and builds a “culture of compliance.”

Maybe one of the most important contributions of the discharge fee program in Colombia is the change in attitudes towards economic instruments, evident in the two following statements by Vice Minister of Environment Fabio Arjona Hincapié (World Bank 2002):

The idea that developing countries cannot use economic instruments ... negates the possibility, the assumption that competitive forces exist in developing nations.

In Colombia's 23 years of experience with command-and-control mechanisms, little pollution control resulted despite very high costs of pollution reduction. Using a broader range of economic instruments is a fundamental change in the signals given to the industrial community.

Voluntary Participation in Emissions Control: Mexico

Voluntary environmental agreements (VAs) have become quite popular in Latin America over the last decades⁶. For example, in Mexico, 10 such agreements involving some 600 firms were signed during the 1990s. In contrast to industrialized countries (where such regulation is used to encourage firms to over-comply with mandatory regulations), environmental authorities in developing countries generally use VAs to shore up the weak enforcement of mandatory regulations resulting from, among other factors, limited public support for environmental protection, weak regulatory institutions, and production dominated by hard-to-monitor small-scale firms (Blackman and Harrington 2000).

VAs include negotiated agreements between environmental regulators and industry associations in specific polluting sectors or geographic areas (negotiated environmental agreements) as well as public programs with pre-established rules to which firms are invited to participate. In this section, we

focus on the case of the Clean Industry Program, Mexico's flagship national voluntary environmental regulation.

The Clean Industry Program is an auditing program created in 1992 by the Federal Environmental Attorney General's Office (PROFEPA). It was designed to leverage PROFEPA's limited resources by shifting some of the burden for monitoring onto the private sector. For that purpose, plants that volunteer to join the program must pay for an environmental audit by an accredited private sector inspector. The audit determines which pollution control and procedures plants have in place and which additional procedures are required to fully comply with the existing regulations. Plants agree in writing to correct all the violations by a specific time; in exchange, PROFEPA agrees to not penalize the plant until that date has passed. Once the agreement is fulfilled, the plant is awarded a Clean Industry certificate that exempts it from regulatory inspections for two years—a "green" badge it can use in marketing campaigns. Notice that the program provides a basket of incentives including both "carrots" and "sticks." The carrots are given by the enforcement amnesty and the Clean Industry Certificate, while the stick is the threat of enforcement of mandatory environmental regulations for plants that do not join the program. This combination has attracted a lot of firms to participate: the number of participating plants grew from 77 in 1992 to approximately 3,500 in 2005.

Official records and statistical analyses (Blackman et al. 2006b) suggest that the Clean Industry Program primarily attracted dirty firms who signed the agreements in order to eliminate deficiencies and be ultimately awarded Clean Industry certificates. Indeed, firms that were inspected and fined were twice as likely to participate in the program than those that did not face any type of regulatory action. This evidence runs very much against evidence found in industrialized countries, where VAs have been shown mainly to attract clean firms. Such firms have a clear incentive to "free ride" on the program since they can obtain significant benefits—positive publicity and preferential treatment by regulators—at a very low cost.

Although there is no clear evidence about the effects of the Clean Industry Program on environmental quality, the fact that a large number of dirty plants joined the program—and ultimately improved their environmental performance—can be interpreted as a positive preliminary indication that the program generated environmental benefits.

Some important conclusions can be drawn from this experience:

- VAs can complement poorly performing mandatory regulation in developing countries. Public-private collaborative agreements can work to induce noncompliant firms to correct violations while filling the gap in regulatory constraints that limit pollution control efforts.
- However, VAs cannot substitute for conventional regulations. The success of the Clean Industry Program is quite correlated to the threat provided by mandatory regulations: plants join the program in order to escape penalties.

- Although VAs can potentially shore up the weak enforcement of mandatory regulations due to limited resources, some environmental regulatory infrastructure is still needed. Clear regulations, regulatory authorities capable of enforcing the regulations, and the availability of pollution control infrastructure affect the credibility of the threat and thus the incentives to participate in the program.
- Finally, disseminating information about firms' environmental performance to the public at large seems to boost the effectiveness of voluntary agreements.

Tradable Permits in Chile

Many donors and advisors have promoted the use of market-based instruments as the key to more effective environmental protection in the developing world. However, less developed countries have had rather limited experimentation with mechanisms such as tradable permits. Santiago, Chile was one of the first cities outside the OECD to implement a tradable permit program to control air pollution. In spite of some institutional capacity constraints, the free market environment of the Chilean economy, the general acceptance of air pollution property rights by polluters, and serious air pollution problems led the environmental authority to introduce a cap-and-trade program in 1992. This program was intended to reduce emissions of particulate matter coming from large boilers, which at the time accounted for more than 40% of total point source emissions (Coria and Sterner 2010).

Santiago's experience shows us the challenges of designing successful trading programs in less developed countries at early stages of institutionalizing environmental protection. At the time the program was implemented, no environmental institution was capable of managing the trading program; a new governmental office was created for that aim. The Program of Control of Emissions Coming from Stationary Sources, PROCEFF, under the Department of Health (SEREMI, *Secretaría Ministerial de Salud*), was given the responsibility of developing a comprehensive inventory of pollution sources and their historical emissions, allocating and keeping an updated record of permits as well as monitoring and enforcing emissions. Within a short time, the first general environmental laws were passed, and in 1994 the National Environmental Commission (CONAMA) was created to coordinate all governmental offices involved with environmental jurisdiction (for example, the departments of transport, economy, and fisheries) and to design new policies to deal with pollution problems. Since then, CONAMA has promoted implementation of additional trading programs for other stationary sources and pollutants. The actual implementation and management of these programs has, however, remained under SEREMI. The fact that institutions and actual regulation evolved so quickly—in some cases simultaneously with

or even superseding legal bases—may have complicated implementation. Trading is officially “recognized” as a policy instrument by the law that created CONAMA. However, the law did not specify the allocation mechanisms, duration, or other characteristics of the permits schemes. Before this law, there was just a Supreme Decree, rather than a law, which established a specific program for large boilers. Although the large boiler decree was passed in 1992, because of limited resources the regulator concentrated its regulatory activity on the completion of the inventory and the allocation of permits. It did not track trading activity until the process was completed in 1998 (Coria and Sterner 2010; Coria et. al 2010).

Grandfathering the permits prompted a rent-seeking behavior that allowed the regulator to more readily identify sources that were not in the original inventory, but it also created incentives for false reporting, particularly given the poor historic record of sources and emissions (Montero 2002). In fact, the lack of accurate information led the new authority to far over-allocate permits, creating a significant excess number of permits in force and preventing the market from fully developing, in the sense that many sources relied on autarkic compliance instead of participating in the permits market. As the program progressed, the environmental authority realized that the initial allocation was too generous and reduced the number of permits in force by reducing the permits granted to existing boilers; it also increased the offsetting rate—that is, the number of permits sources need to buy in order to emit particulate matter. These regulatory interventions have affected the tenure of emission permits and hamper trade, since firms prefer to keep permits in excess instead of selling them in order to deal with the uncertainty regarding eventual new changes.

The design of the scheme has shown to be inadequate in three ways. First, requirements for prior regulatory approval (not to mention the limited resources devoted to SEREMI) explain the delayed response of the regulator to the offsetting requests by polluters. Indeed, the average period required for a transaction to be approved in the period 1998–2003 was about 20.5 months. Fortunately, the number of months the transaction process lasts has been trending downwards over time.

Second, institutional and legal constraints have prevented SEREMI from setting fixed and automatic monetary penalties for emissions violations, affecting the capability of the regulator of enforcing the policy. Instead, noncompliant firms have faced sanctions imposed through administrative procedures. Sanctions are decided on a case-by-case basis, and they might include a note of violation as well as a wide range of lump-sum monetary sanctions. As a consequence, almost 30% of large boilers included in the program have not met their emissions cap obligations at some point during the period 1997–2007.

Finally, lack of expertise and resources led SEREMI to under-provide information to the market. Information about emission permits in force,

actual emissions, penalties, noncompliance, and reports of early trades and prices have not been publicly available. This has increased transaction costs since price information required by firms to make investment and compliance decisions is not easily available. In addition, it has affected the credibility of the scheme, because when penalties for noncompliance are unclear and are at the regulator's discretion, they can easily be manipulated. This is particularly the case when noncompliance is explained partially by the delayed answer of the regulator to an offsetting proposal by firms.

Yet in spite of those weaknesses, the aggregate cap on emissions has been met, and trading activity has increased through time. However, it is likely that full cost-effectiveness has not been achieved.

A number of design modifications could have substantially improved the efficiency of the Santiago system:

- better records and measurement of emissions;
- more certain tenure over the permits;
- a simple and stable scheme of penalties;
- transparent and evenly applied enforcement;
- avoiding rules that hamper trade, such as offset rules that provide a bias against trade;
- enhancing public access to information about trading and compliance.

From this experience we see that, as in Colombia, no clear reasons lead us to believe that developing countries cannot benefit from the additional flexibility that tradable permits confer over more inflexible regulations. One should keep in mind that it took the U.S. some three or four decades of experimentation to learn how to design the institutions for a trading scheme. The Chilean scheme compares quite favorably with all the early U.S. programs, and even to the European emissions trading scheme (EU ETS) which, in spite of being launched long after the Chilean scheme, has roughly speaking the same number of flaws related to overallocation and a lack of clear and consistent penalties. Thus, one might say that this experience demonstrates that a middle-income country such as Chile is very capable of implementing this type of scheme even if much work remains before the design is really satisfactory. Considering the urgency of improving the environment in many of these countries, it is important to use a whole range of potential instruments.

Differentiated Electricity Tariffs: Mexico and Zambia

The cost of providing infrastructural services such as water or electricity depends crucially on the physical supply and the number and consumption level of individual consumers. The density of consumers in urban areas makes the necessary investment per capita for their services much smaller than in

rural areas. The cost of supplying a certain customer does not consist of only the (long-run) marginal cost for bulk electricity (or water) but also the costs for local distribution and associated costs of metering, monitoring, administration, and so forth. These costs are all essentially fixed, which means that low-volume consumers are much more expensive to supply than high-volume consumers. The supply curve thus slopes downward for the relevant interval.

In setting tariffs, several factors must be considered together. Conceptually, tariffs can be described as regressive, progressive, or unitary depending on whether they rise, fall, or remain constant with rising consumption (see Figure 21-2). In recognition of the fixed costs of providing a service, one can argue that high-volume consumers should pay less; the tariff system then follows the supply curve, which promotes efficiency because it reflects actual economic costs for the supplier. We call this kind of charge an efficiency tariff. Each customer pays the costs incurred by the supplier, and thus the supplier has reasonable incentives to provide service to all customers. On the other hand, this tariff may be seen as regressive.

At the other end of the scale, the progressive tariff essentially takes income distribution or ethical (political) considerations as its starting point. It provides services (e.g., electricity or water) to poorer (lower-volume) consumers at subsidized rates and charges richer (higher-volume) consumers increasingly higher rates. This kind of charge may appear to be the “greenest” tariff construction and most appropriate in reducing wasteful consumption: the minimum (necessary) consumption is cheap, but the more wasteful uses are

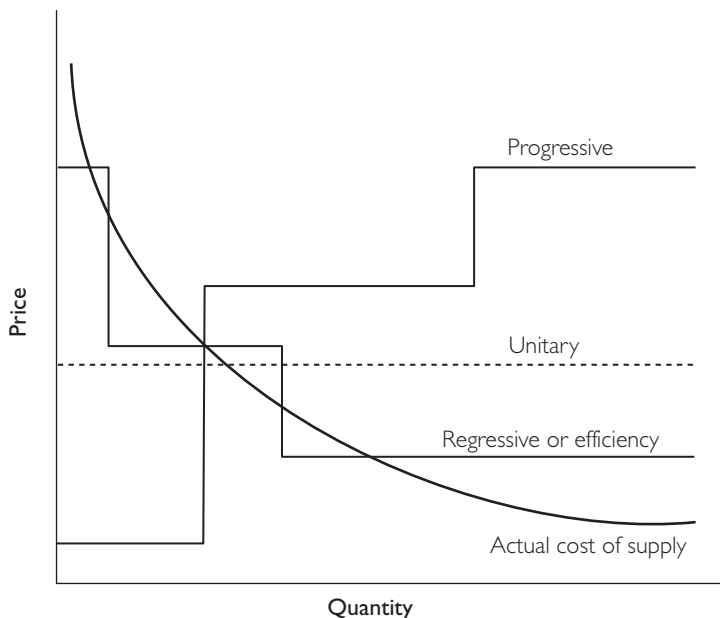


FIGURE 21-2 Different Kinds of Tariffs

discouraged by a high price. However, when analyzed carefully, the issues are complex, and the progressive tariff is not necessarily attractive—neither for the poor nor the environment.

The unitary tariff is in a sense “progressive” in that it provides subsidies to low-volume consumers and taxes high-volume consumers relative to the actual costs of supply. The unitary tariff also is simpler to implement than other kinds of charges because cheaper meters and administrative routines can be used, and less incentive for cheating exists.

It is a misunderstanding that progressive tariffs are needed for resource conservation. Conservation requires a correct value of the resource rent. For individual consumers, this value does not rise. As society uses more of a certain resource, the rent may rise, but this increase should then be reflected in rising tariffs for everyone over time—not necessarily a progressive tariff schedule for each consumer. Problems arise when the appropriate rent level is so high that poor people cannot afford their small but vital consumption.

The progressive tariff also makes the poor customers unprofitable for the supplying company. In principle, cross subsidies might be possible in some contexts (e.g., the company might make its profits from the rich, high-volume consumers and subsidize the poor, low-volume consumers) but unlikely on a large scale. Instead, the poor consumers end up receiving second-rate services—or no service at all. For instance, when an accident cuts off the electricity supply to a poor community, the electricity company has a perverse incentive to not restore power; the company will lose money as soon as the problem is fixed. As a result, power outages are more or less permanent in some areas (Köre and Widepalm 1993). This is not a good incentive structure. In fact, the general state of public goods supplied to poor communities in many developing countries illustrates this point all too well. The water is often of poor or uncertain quality, and supply may be unpredictable; electricity outages are frequent and bus services inadequate. Poor people want public services to be cheap, but their primary concern is that the services should be functional. If tariffs are such that the supply companies or transportation organizations do not manage to break even, then the people will not get the services they need.

Progressive tariffs may even disfavor the many families who share an apartment, house, or room with only one service contract, because the total consumption levels of all these families is too high to be charged the low-volume tariff. On the other hand, some high-volume customers are able to cheat the system by acquiring several contracts in the names of different family members, and if each of these separate contracts has sufficiently low consumption, then they all will be charged the minimum tariff.

With efficiency tariffs, the requirement is clear: a mechanism to distinguish high-volume consumers from low-volume consumers. Usually, this distinction is accomplished by the use of a two-part tariff, so that the low energy price comes with a fixed meter or ampere charge that is related to the maximum capacity installed. This fixed charge is prohibitively high for

TABLE 21-2 Some Electricity Tariffs in Mexico, 1993

<i>Tariff level</i>	<i>Electricity consumption</i>	<i>Rate (N\$/kWh)</i>
1: Residential	025 kWh	0.06
	25–50 kWh	0.09
	50–75 kWh	0.11
	75–100 kWh	0.13
	100–200 kWh	0.15
	>200 kWh	0.47
2: Small commercial/industry (<25 kW)	050 kWh	0.32
	50–100 kWh	0.41
	>100 kWh	0.45
3: Large commercial/industry (>25 kW)		0.22
9: Irrigation	5,000 kWh	0.10
	510 MWh	0.12
H-M: High tension (>1,000 kW)		0.13

Notes: N\$ = nuevo peso; kW = kilowatt; kWh = kilowatt-hour; MWh = megawatt-hour.

Source: Comisión Federal de Electricidad 1994.

low-volume customers, who are thus forced to reveal themselves or their type and thereby be charged the higher energy tariff (but lower fixed costs). (This simple example illustrates the revelation mechanism discussed in Chapter 10.)

Actual tariffs for water and electricity in various countries vary considerably from the progressive to unitary or efficiency models. For many years, Mexico has favored progressive tariffs, so large industries and high-volume consumers pay much more for electricity than low-volume consumers do. Table 21-2 shows a selection of electricity tariffs for Mexico in 1993, after some of the heaviest subsidies had been removed but considerable cross subsidies remained. If the price of high-tension power for large industries was 0.13 pesos/kWh, then the cost of supplying residential electricity was probably much higher. Yet the low-volume consumers paid less than half this rate for small quantities of low-tension power (0.06 pesos/kWh), while medium-volume consumers paid somewhat more and small industries paid more than five times as much. Irrigation received large subsidies, whereas commercial and small-industry tariffs were hardly subsidized. One result of this policy is that many low-income Mexican households still lack electricity because supplying them entailed large losses to the electric company for many years. Since the late 1990s, these cross subsidies have been removed or at least reduced, and low-tension tariffs have higher energy charges than high-tension tariffs.

In Zambia, where the mining industry creates a large-scale demand for electricity, millions of rural and even urban inhabitants live without electricity and depend on biomass fuel for household energy.⁷ The country has considerable hydroelectric potential, but most of it is still untapped, partly

for lack of funds in Zesco, the state electric company. For many years, Zesco had a monopoly on electrical service but was not allowed to charge prices that covered long-run marginal costs to most of its customers. Before 1994, at least 19 electricity tariffs were used, 12 of them for unmetered use by households, and all low (Mulenga 1999). For instance, low-volume household customers could choose connections with load restrictors of 1 to 15 amperes. A small household might have had a 2-ampere connection (Tariff LC) and would have been charged a fixed fee of 15.7 kwacha per month in 1992. In contrast, under Tariff E1, intended for somewhat higher-income households (and higher-volume consumers), the energy charge was 17.35 kwacha/kWh. Supposing the latter had been a true reflection of cost, then the households with the unmetered 2-ampere connection would have been paying less than 1 kWh/month—the equivalent of using one lamp for 10–15 hours. The unmetered customers were getting a really good deal on electricity service, but most people who wanted a connection could not get one.

A somewhat more pragmatic policy has been followed for large industrial customers, which can choose tariffs that have lower marginal costs and higher fixed or capacity charges. However, the tariffs are updated infrequently, and with high inflation, their real value tends to fall below the long-run marginal cost (see Table 21-3). All the tariffs are still so low that none covers marginal cost. Furthermore, some tariffs carry an implicit subsidy that is larger than others. The commercial Tariff C2 caters to large customers and thus has a low marginal cost, but its price is higher (US\$0.0586/kWh) than the commercial Tariff C1 (US\$0.0436/kWh) for small consumers. This kind of tariff setting breeds corruption because the subsidized good inevitably is in great demand, and those in charge of meeting demand probably can grant favors to certain people. Moral character or culture does not necessarily determine the extent of corruption in a

TABLE 21-3 Electricity Tariffs and Long-run Marginal Cost of Electric Supply by Tariff, Zambia, January 1998

Consumer category	Tariff type	Price charged		Estimated cost (LRMC)	
		Capacity (US\$/kV-A)	Energy (US\$/kWh)	Capacity (US\$/kV-A)	Energy (US\$/kWh)
Residential	R2		0.033		0.08
Commercial	C1		0.044		0.15
Commercial	C2		0.059		0.09
Industrial	MD1	6.25	0.035	12.50	0.04
Industrial	MD4	8.33	0.016	11.66	0.04
Social services	H1		0.042		0.09

Notes: LRMC = long-run marginal cost; kV-A = kilovolt-amperes; kWh = kilowatt-hour. The LRMC does not include the cost of metering, billing, and marketing.

Source: Adapted from Mulenga 1999.

society; the determining factor is more likely to be technical matters that could easily be resolved. Political economy is very important here: there is a risk that tariff setting is influenced by the demands of the “insiders” (who already have electric connections) with insufficient regard for the interests of the outsiders, who will not receive electricity service unless it profits the supplier.

Information Provision and Institutional Capacity: Indonesia

Indonesia epitomizes many environmentalists’ worries about economic growth⁸. Economic growth has been spectacular—exceeding 10% for long periods—but it has been based on a drastic exploitation of natural resources such as oil and timber. Estimates of the country’s real (net) growth (corrected by subtracting for degradation of natural capital) are much smaller, but still impressive (e.g., Repetto et al. 1992).

The country is also known for an authoritarian government in which the family and associates of President Suharto (1967–1998) wielded considerable power and influence not only in politics but in business and administration as well. Despite the existence of a regulatory regime in the early 1990s, the attempts by BADEPAL—the local environmental protection agency—were not very successful. Many of the owners of polluting industries were powerful, and BAPEDAL was a relatively small and weak organization. No fines were ever assessed, and the authorities lost 90% of the environmental and health court cases (García et al. 2009). The risks of ineffective monitoring, weak enforcement, and even corruption were obvious, at least if the program were to seriously threaten the interests of the ruling elite.

BAPEDAL eschewed the idea of regulations (which, in principle, already had been tried with no effect for many years) because it believed it had no chance of success against industry in trying to enforce closures, fines, or other penalties. The agency was also wary of a system of environmental charges, in part because they create a situation of negotiation between the polluter and individual officers of the agency, which might bring incentives for bribery. Instead, they chose a rating or labeling system: the Program for Pollution Control Evaluation and Rating (PROPER).

What is distinctive about PROPER is the care and attention put into all details of the design of the system. Fundamentally, it is a system for emissions reporting, evaluation, and control of emissions reports and ratings; it also offers assistance and advice to firms. Each industry is graded, and ratings are based on several parameters in the reports. A good deal of effort was spent on designing the parameters. The purpose was to have an appropriate set of parameters: not so many as to make reporting burdensome and alienate the firms, but not so few that the reports would be easy to falsify. By experience, BAPEDAL officials knew that stoppages, for example, cause the excess release of certain, although not necessarily all, pollutants. Such stoppages are

related to other parameters that the firms report: production, electricity use, raw material consumption, water, and so forth. Therefore, the PROPER team collected many indicators and developed small but sophisticated programs for checking the correlation among the various data collected.

Engineering studies and programs have given the PROPER inspectors a valuable tool: knowledge. These inspectors must be well prepared for inspections and discussions with the firms in order to gain the firms' respect. They also must be perceived as helpful, impartial, and influential. In detecting a release or a problem, they must not be perceived as being "out to get" the company. Confidence, respect, and communication are vital. Many problems found during inspections can be resolved by the firms, and in many cases, the PROPER team has the expertise to help. Inspectors acquire and improve their expertise because firms respect them and share technical knowledge with them.

In addition, the inspectors must be seen as incorruptible. Even one incidence of corruption can ruin the reputation of this kind of system. The process of collecting, evaluating, and rating emissions data is highly automated in a formalized computer program, and a staff of fewer than a dozen well-trained people runs the whole scheme. The rating and evaluation procedure is also transparent, so firms know exactly what they have to do to improve their grades. Meanwhile, other agencies, competitors, the press, and the public can verify and control not only the companies but also the work of the BAPEDAL regulators. Some neighboring communities affected by heavy pollution have challenged ratings (and thus a firm's emissions data), and in one case, a community succeeded in changing the rating of a firm that had cheated to receive a Green rating (when it should have been Black).

To understand at least partly the complexity of just monitoring, consider Figure 21-3, which shows data for one measure of effluent from one Indonesian factory. The data for this particular industry show a clear response to the PROPER program. Such diagrams may be difficult to interpret. Note in particular the daily variation—fairly typical here—with corresponding variation by the hour or minute, making actual measurement quite an art. If sampling is the chosen tool, then what mechanisms must be in place to guarantee that the sampling is representative? How can the system ensure that the company is not releasing massive amounts of emissions while the measurement system is inactive? Furthermore, Figure 21-3 shows only one measure (BOD concentration), but other measures of pollution include more detailed counts of certain emissions. Regulations must determine the percentage of observations that are reasonable under a compliance standard. If only a few measurements are taken, then the risk is high that the measurements are unrepresentative. The greater the number of observations, the smaller that risk—but then decision algorithms must be created to distinguish between allowable anomalies and systematic violation. If too much power of judgment were left to individual inspectors, then transparency would diminish and the risk of corruption (or of being suspected of corruption) would surface.

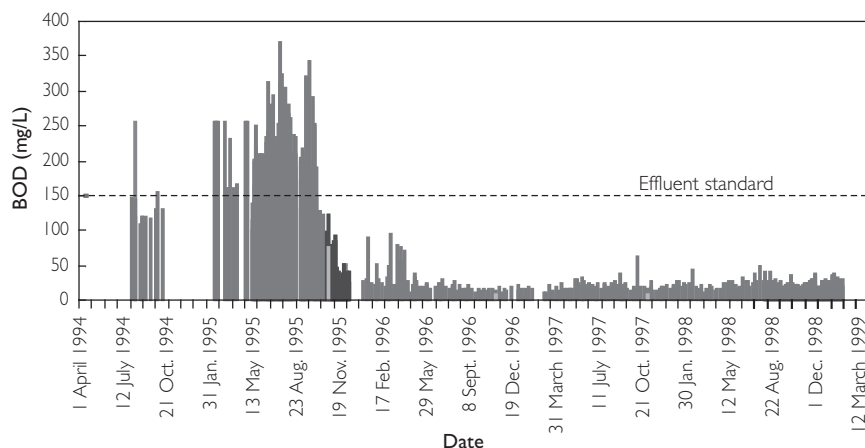


FIGURE 21-3 Daily Variations in Pollution and the Effect of PROPER in Indonesia, after 1995

Note: Effluent standard is marked at 150 mg/L.

Labels were carefully chosen to be simple, clear, and understandable to citizens who have little or no prior knowledge or understanding of technical and pollution issues. For simplicity, a color-coding scheme was chosen, but for coherence and credibility, the colors have precise meanings. The color categories are described in Table 21-4. The entire system builds on earlier regulation. Firms that simply comply with environmental regulations are Blue. Firms that are proactive and significantly exceed the environmental standards can be awarded a Green status, and the world-class Gold status has yet to be achieved by any firm in Indonesia. Firms that fail to meet minimum standards are Red or, in cases of significant environmental damage and no effort at abatement, Black.

Information disclosure has been handled carefully. The PROPER team started by evaluating 187 large and medium-sized polluting plants in 1995. Initial ratings indicated that two thirds of the companies were noncompliant and would receive Red or Black ratings, showing how poorly regulation—which

TABLE 21-4 Color Labels Used in PROPER

Rating	Technical requirements
Gold	World-class clean technology Waste-minimization and pollution-prevention efforts
Green	Above legally required standards for environmental protection Good maintenance and environmental work
Blue	At legally required standards for environmental protection
Red	Below legally required standards for environmental protection
Black	Serious environmental damage No pollution-control effort

TABLE 21-5 Change in Firms' Ratings Due to PROPER Disclosure in Indonesia

<i>Rating</i>	<i>June 1995</i>	<i>Dec. 1995</i>	<i>Dec. 1996</i>	<i>Change (%)</i>
Gold	0	0	0	0
Green	5	4	5	0
Blue	61	72	94	+54
Red	115	108	87	-24
Black	6	3	1	-83
Total firms	187	187	187	

Source: Data from BAPEDAL (Indonesia Environmental Impact Management Agency) and World Bank 2000.

formally had been in place for a long time—had worked. The PROPER team then took two crucial steps. First, it went to considerable trouble to verify data, vetting the ratings at many levels, holding discussions with representatives of industry and NGOs, and having the ratings approved by not only the minister of environment but also the president. The knowledge that the president had seen and approved the ratings was crucial in giving them credibility and legitimacy. Second, the team went to great lengths not to antagonize industry, because it was crucial that the PROPER team was not to be perceived as “anti-business.” They therefore started by informing all the industries and giving those with poor ratings six months to improve before public disclosure. Already, this first step had considerable effect. Then, they disclosed the names of only the handful of firms with really good ratings, which were congratulated in a highly publicized event with Vice President Tri Sutrisno.

In December 1995, full disclosure of all the ratings started, industry by industry, to receive maximum press coverage during a fairly extended time period. The changes in rankings from June 1995 to December 1995 and December 1996 are shown in Table 21-5. Response was immediate during the first six months of the program—even before disclosure—particularly among plants rated Black or Red. The incentive to avoid the disgrace of a bad grade appeared to be the strongest factor in increasing firms' ratings. All but one of the Black ratings improved, and 24% moved up from Red to Blue. The most important sign of progress was that in the first 18 months, overall effluents from the first 187 firms inspected decreased by 43%.

Another aspect of BAPEDAL's PROPER strategy was the gradual inclusion of more firms. Concentrating its efforts on the most polluting industries first, the agency could target approximately 20% of the pollution coming from just 1% of the firms. Moving on to 10% of the plants would cover roughly 90% of pollution, and this effective strategy concentrates inspection efforts on firms where the environmental effect per control dollar will be highest.

Several possible mechanisms appear to be essential to the success of the PROPER scheme (see Chapters 2, 4, and 10). The most relevant are the following:

- **Media and reputation:** Firm managers feel a strong sense of civil responsibility. Indonesia has a strong culture of shame, so managers generally try to avoid bad publicity.
- **Consumer reaction:** All other things equal, a firm's reputation may influence consumers' purchasing decisions. This response is likely only in cases when the number of producers gives consumers an array of choices. One might think that this mechanism works only when the industries produce consumer products, but in fact, reputation also sometimes appears to be important for firms that produce inputs for other firms.
- **Worker reaction:** Employees, like managers, must take pride in their workplace to be motivated to do a good job. In extreme cases (high levels of pollution), employees may hesitate to work in a particular plant or demand a higher salary to do so. This factor presupposes that laborers have some choice in employment, so it applies mainly in an economy that offers reasonable levels of employment.
- **Investor reaction:** Like consumers and workers, investors may make investment decisions based on ethical, environmental, or other sensitivities. Even if they do not, they may believe that regulators or communities will react to environmental problems and thus may worry about the hidden liability risks of noncompliant firms.
- **Community reaction:** Local, regional, and national communities may react to pollution. Public disclosure is perhaps the instrument that makes this kind of reaction most likely, because it does not depend on the action of a few environmental protection agency officials. It can be influenced by NGOs and citizens who have access to information and can thus monitor the actions of not only the firms but also the regulators.

It is difficult to judge the relative importance of all these factors in the success of the PROPER program. Foreign-owned firms, which seem to be more sensitive to local and foreign media attention and subject to stricter public scrutiny, and firms located in the most densely populated areas—and therefore subject to a greater potential for community pressure—were more sensitive to reputation pressure (García et al. 2009). Perhaps the most important mechanism that encourages success is the simple provision of correct, reliable, technical and environmental information to managers (Afsah, Blackman, and Ratananda 2000). In this respect, PROPER is fulfilling the role of an environmental audit. Firms that apply for international licensing such as ISO 14000 also are keen to use information in this way.

PROPER appears to have an advantage for the kind of quick, broad-based cleanup effort that constitutes the first stage of environmental management. To deal with the diversity of pollutants and processes, the tools for monitoring, licensing, and information must encourage good management practices. In these contexts, charges and tradable permits are not easily applicable. In some of the formerly planned economies in Eastern Europe, long and detailed lists

of chemicals with specific charges or fees were attempted. This scheme did not work well and is likely to be untenable, partly because of the costs of properly supervising such a complicated procedure. In contrast, the cost of the PROPER scheme was modest (approximately US\$100,000) for the first 18 months. Almost 200 plants were rated, so the average cost was approximately US\$500 per plant, which presumably could be financed through license fees. Given the pollution reduction of 40%, the PROPER program was cost-efficient. Unfortunately, the severe economic crisis in Asia more or less killed the program before it became fully institutionalized, and little of it remains.

Several other countries have adopted similar programs. Most advanced is the Philippine EcoWatch program, which has many similar features, including the color-coding system. Under EcoWatch, in the initial evaluation of 52 plants in the Manila area, 48 were found to be noncompliant (with Red or Black ratings). Failure of conventional regulation was the main motivation for the labeling scheme. Within just over a year, the number of plants rated Blue increased from 8% to 58% in 1998.

Mexico, Papua New Guinea, and Colombia also have begun public disclosure systems, and a handful more countries have announced similar intentions or interests. In Colombia, the program will complement the charges discussed above. In India, an NGO has started the Green Rating Project to monitor the environmental performance of large Indian firms (Kathuria and Haripriya 2000). We turn next to India.

Two-Tier Pollution Regulation: India

The Golden Corridor through the Indian state of Gujarat is a collection of pollution hot spots⁹. Gujarat, located in west India on the border with Pakistan, has become a “toxic haven,” as reflected by the relocation of the hazardous ship-breaking industry from the industrialized world to Asia. More than one half of the world’s ships are dismantled in one Gujarat town alone (Alang). This hazardous task involves exposure to large quantities of asbestos, arsenic, residual oils, paints, and other toxins. The breakup is not done in shipyards; the vessels are simply driven up onto the beach, stripped, and cut up with hand tools (such as torches) by migratory workers who lack protective clothing and accessories. One of the worst sources of pollution in this area is a group of chemical industries located in the Ankleshwar Industrial Estate (AIE), not far from Alang.

During the past 30 years or so, many environmental laws have been passed in India. These laws provide for the use of command-and-control measures, economic instruments, and institutions that facilitate participation in environmental management. The Environmental Protection Act empowered the government through the Ministry of Environment and Forests to create central and state pollution control boards intended to prevent, control, and abate pollution. In Gujarat, the highest environmental authority is the

Gujarat Pollution Control Board (GPCB). It can demand information from any industry, issue fines, and even order closure. The most important legislative provision that has made polluters more conscious of and susceptible to legal action is public interest litigation. At least on paper, Indian authorities have a wide array of instruments at their disposal, including formal standards, varied frequency of inspection, subsidies, taxes, and fees as well as information disclosure, the imposition of polluter liability, and zoning.

Industrial Estates in India

A large share of organized (polluting) industries in India are located in industrial estates, which are zoning tools for industrial dispersal and the industrialization of “backward” areas. This emphasis on zoning potentially could include a strong focus on adapting the nature of industry to the carrying capacity of the ecosystems, or at least some consideration of infrastructure such as water supply and waste disposal. At present, nearly 1,000 industrial estates exist in India.

Industrial estates may play an important role in the regulatory process. They have some form of democratic or cooperative structure, and pollution from within their boundaries can be considered nonpoint-source pollution to some extent. For regulators and residents outside the estates, it is difficult and expensive (although not necessarily impossible) to monitor individual plants, but aggregate pollution from an industrial estate is noticeable and attracts attention that may lead to bad publicity, complaints, and even plant closures. A two-tiered structure of monitoring may arise because more detailed monitoring is cheaper and easier for insiders (i.e., in the estate). In addition, industrial estates have some incentive to regulate and discipline their own members because all the member industries in an industrial estate can be hurt if the estate’s reputation is tarnished. It is therefore relevant to refer to the literature on common property resources and to the theory of nonpoint-source regulation (see Chapters 23 and 10).

Although pollution control boards are formally mandated to monitor individual plants in an estate, given the enormity of the task and the costs involved, they may monitor only the aggregate emissions from the estate if the estate takes the initiative to monitor its members. The intermediary position of the estate is tricky, however. Its only instruments include the provision of infrastructure (partly subsidized by central authorities) and voluntary systems of fees and sanctions. It holds no formal enforcement power. In the following subsections, we discuss the efficiency of these instruments.

Policymaking and Mediation: AIE

The AIE is one of the largest chemical estates in India. It covers an area of 1,605 hectares and houses more than 400 chemical plants. These units manufacture

more than 25% of Gujarat's (5% of India's) output of pharmaceuticals, chemicals, pesticides, dyes, and dye intermediaries. The Ankleshwar Industries Association (AIA) has estimated that its members generate between 250 million and 270 million liters of liquid waste daily and approximately 50,000 tons of solid waste annually. As a direct consequence, villages on the periphery of the AIE must bear heavy costs in the form of water and air pollution. The major local water body, Amlakhadi Creek, is completely void of biological life. Neighboring villages have suffered from the poor quality of the groundwater, which allegedly has contributed to the death of some cattle.

During the past few years, the AIE and other estates have started to take some preliminary steps toward cleanup—perhaps in part because of easier access to public information, public interest litigation, and highly publicized tragedies such as that in Bhopal, which seem to have made polluters more conscious of the risk of legal action. The estate also is situated along the Bombay–Delhi railway route, highly visible to passersby.

The AIE has used several instruments to coerce its members, including provision of information, direct regulation (of emission standards), effluent fees, and fines. These efforts were partly related to the creation of a common effluent treatment plant (CETP) for small-scale industries and a central land-fill for sludge and solid waste. These facilities were subsidized by the government but operate on the polluter pays principle (i.e., each unit pays according to pollution load and quantity of waste generated). An additional and presumably important policy instrument for AIE is publicity. In 1989, AIA created the Ankleshwar Environment Preservation Society (AEPS) to take a more proactive stance. The two aims of the society are to assist industries in disposing of wastes and liquid effluents and in controlling air pollution, and to set up a laboratory for testing samples of air and liquid effluents. The society has purchased monitoring equipment, offered educational programs, and planted trees. It was one of the first organizations in the country to obtain tax concessions for pollution-control projects.

One of the activities of the AEPS is the monitoring of effluents and emissions from chemical industries in the AIE. If the pH, COD, and other values of samples are found to be in excess of GPCB standards, then the AIA environment committee calls the concerned units for discussions. After a couple of notices, the AIA levies a financial penalty or other sanction. These penalties are carefully graduated and increase proportionally with the frequency and severity of the offense, pollution, or method of discharge (some forms of illegal discharge outside the designated area are particularly reprehensible). For example, the penalty structure for the illegal pumping of acidic effluents into drains is listed in Table 21-6; similar disposal outside the drainage system carries even stiffer penalties. Other infractions against COD or other rules have different but similarly strict penalties.

After repeated infractions by a unit (or when penalties are not paid), the AIA's last resort is to report that unit to the GPCB. The existence of this threat

TABLE 21-6 Penalties Imposed by the Ankleshwar Industrial Association for Illegal Pumping of Acidic Effluents into Drains at the Ankleshwar Industrial Estate in Gujarat, India, 1998

<i>Offense</i>	<i>Penalty (rupees)</i>	
	<i>Small enterprises</i>	<i>Medium and large enterprises</i>
First	2,500	10,000
Second	5,000	20,000
Third	10,000	40,000
Fourth	Report to the Gujarat Pollution Control Board	

is a decisive mechanism. The AIA will not overreport to the GPCB, because overreporting is a sign of failure on its part. The AIA also hesitates to report because reports of one unit may lead to criticism of the entire industrial estate. Additionally, inspection of one unit by GPCB may trigger inspection of other units in the estate. All of these reasons could explain why, by early 1999, the AIA had reported only three perennial defaulters to GPCB. The way that GPCB reacts also will be important for the success of the AIA “voluntary” fines in future monitoring and detection situations.

From 1996 to 1999, several units were subjected to mild penalties for violating GPCB standards. Penalties levied and recovered from polluters are listed in Table 21-7. Although the amount of penalty levied is increasing nominally, the rate of collection over time appears to be decreasing. This trend indicates an apparent erosion of AIE authority. However, the number of observations is too few to be certain, and the incomplete figures for 1998 are particularly uncertain.

The decrease in the percentage of collected fees and levied penalties is attributable to the fact that AIA and AEPS are voluntary organizations and not authorized by law to collect fines, so defaulting units may not feel obliged

TABLE 21-7 Fines Levied and Collected by Ankleshwar Environment Preservation Society/ Ankleshwar Industrial Association at the Ankleshwar Industrial Estate in Gujarat, India, 1996–1999

<i>Year</i> (1)	<i>No. of fines levied</i> (2)	<i>No. of fines collected</i> (3)	<i>Collection rate (% of no. of fines)</i> (4)	<i>Total amount of fines levied^a</i> (5)	<i>Amount collected^a</i> (6)	<i>Collection rate (% of amount collected)</i> (7)
1996–1997	150	36	24	6.9	5.5	80
1997–1998	196	38	19	9.8	6.0	61
1998–1999 ^b	186	37	20	15.1	2.7	18
Total	532	111	21	31.7	14.2	44.6

Notes: ^aAmounts in hundreds of thousands of rupees (also called *lakhs*). ^bData incomplete; to December 1998 only.

to pay. Compliance hinges on voluntary participation, but all industries should realize that it is in their common interest to achieve minimum abatement to protect their collective reputation. However, this situation is a classic “Prisoner’s Dilemma” (see Chapter 8), and the rapid decrease in voluntary compliance with the fines may be one example of a move from an initial collaborative equilibrium to a Nash equilibrium (see Chapter 8). After a few firms succeed in getting their fines waived or realize that nothing really bad happens to the 20% of firms that did not pay in the first year, other firms quickly lose their incentive to collaborate.

Rules and institutions must be carefully designed to avoid this kind of problem (Ostrom 1990). People who collaborate to manage common property resources tend to be on a relatively equal footing. They commonly are also linked by history and culture, which creates trust, traditions, and norms. However, among a collection of industries, disparities in size and power may be enormous, and the cultural factor is definitely weaker than in a fishing or farming village.

One of the most important institutions for collective self-governance is a system for monitoring, enforcement, and punishment, not unlike the AIA’s graduated sanctions. Other important criteria concern congruence among duties, obligations, and democratic influence. One might suspect that disparity in size and influence among the large and small plants would be a major difficulty, and anecdotal evidence from Gujarat suggests that many companies suspected that firms that had executives on the boards of the AIA or AEPS would receive more lenient treatment or have their fines waived. However, the collection rate for the total fine amounts (see Table 21-7, Column 7) is much higher than the collection rate of the number of fines (Column 4). These results indicate that the AIA failed to collect the smaller fines, not the larger ones. The main reason probably is that small firms simply have no cash, which suggests that these data do not support an undue influence by the larger firms.

Fines levied and collected serve a dual purpose because they are used to finance recurring costs as well as part of the capital expenses of the AEPS testing laboratory. A decreasing collection rate has a direct bearing on future monitoring. Records of the penalty letters issued for COD violations show that in the first six months of 1998, 64 plants were given a total of 202 penalty letters amounting to total fines of 6 million rupees, twice the actual penalty collected by the AIA in the previous three years. Surprisingly, the owners of three faulting units are members of the executive committee and the environment committee.

Provision of Infrastructure: Effluent Treatment, Landfill, and Pipeline

A crucial component in addition to the monitoring and enforcement system is the provision of local public goods in the form of infrastructure for abatement, treatment, transport, and disposal of wastes. This task is intimately linked with the monitoring task. The estate has set up an effluent treatment

plant (the CETP) and a landfill, and built a pipeline to the sea. These projects warrant some comment.

In principle, the CETP provides primary, secondary, and tertiary treatment and was built exclusively for the small-scale industries, which were believed to have neither the necessary finance nor the know-how to undertake such treatment on their own. The state provided a 25% subsidy for construction costs and subsidized land, but the CETP is run as a private enterprise according to the polluter pays principle. The sludge from the CETP is to be deposited at the waste disposal site with some solid waste—also for a fee. The disposal site was built according to modern specifications to avoid leakage and also runs on a commercial basis (the only subsidy was half the cost of the land needed).¹⁰

A couple of serious flaws are evident. First, at a technical level, the CETP concept is not credible, because the pooling of various chemicals creates an unknown mixture that is difficult to treat properly. If many toxins are present, then microbes will also be killed off, making it impossible to properly treat even the biodegradable fraction—waste that would have been fairly simple to treat in isolation. Tests carried out by Greenpeace show that the treated effluent and the sludge contain the same toxins, casting considerable doubt on the efficiency of the treatment.

Also, serious organizational setbacks have plagued the AIE. Many small firms appear not to be members of the CETP and are suspected of using open drains or other illegal means of waste disposal. Small firms complain about high charges and the inflexibility of the contract. The charges are fixed, irrespective of size, making the fee astronomical for some and insignificant for others. Small firms also complain that dumping fees are not flexible with respect to waste load (which varies with production). Finally, they complain that competitors in other states do not face the same environmental regulation and thus can undercut their prices. In addition, several large firms use the CETP, even though it was intended to be used only by small firms. Apparently, either their own treatment plants are insufficient, or they save costs by using the CETP. This practice may increase the operating costs of the CETP and, ultimately, the fees that the small-scale industries are paying, which aggravates the controversy surrounding this issue.

It is also interesting to note the reasons some chemical firms pointed to for joining the CETP—that is, the indirect benefits of being an AIE member. Membership entitles the unit to a “zero discharge” certificate, which reduces monitoring pressure. Several units specifically said that membership meant fewer visits from GPCB officers. Under the amended Water Act of 1988, the GPCB wields a lot of power; it can strip electricity and water supplies to offending units. Anecdotal evidence indicates that some firms feel obliged to treat the GPCB officers well to avoid any risks, and thus they highly value any reduction in the frequency of their visits.

The Amlakhadi Creek, which is extremely polluted, is 14 kilometers long and empties into the Narmada River. The creek carries effluents from the AIE as well as Panoli and Jhagadia Industrial Estates. It was reinforced in

1995–1996 at an estimated cost of 6 million rupees (which was recovered from the industries) to reduce the spread of effluents to groundwater. However, since the condition of the creek remained unsatisfactory, the AIE environment committee suggested constructing a pipeline to the deep sea for “safe disposal of the effluents.” This project had some support from the World Bank, and although the dumping of toxic wastes into the sea is tragic and might even violate international conventions, it is a local improvement.

Although insufficient, several serious efforts are made to limit pollution in Indian industrial estates, including the provision of infrastructure and the use of various policy instruments. The instruments include the provision of information, direct regulation (pollution standards), fees, fines, and subsidies at various levels. The voluntary efforts of the AIE have led to monitoring of the polluting industries, the collection of some self-imposed penalties, the construction of the CETP and the waste site, and the pipeline to the sea. Monitoring and enforcement appear to be difficult, and the effluent treatment and disposal systems have serious flaws. However, central authorities have grossly insufficient resources, and the AIE may have the potential to play the role of an intermediate principal in a two-tier regulation game. However, experience to date suggests that playing this role is not easy.

Lessons Learned

Three waves of environmental policymaking in developing or transitional countries have been described: command-and-control regulation, economic incentives, and provision of information (Tietenberg 1998). To this excellent and concise characterization we add a fourth category: the construction of institutions for the allocation of rights that are fundamental for any market mechanism.

In practice, many specific institutions are needed, for example, bodies that make rules for liability, courts, and organizations for monitoring and enforcement as well as the news media and agencies that deal with economic and environmental regulation at local, national, and supranational levels.

These three or four categories of policymaking do not necessarily have to be repeated in the same order as they occur in the industrialized countries. They are mutually dependent and must be built up in a fine balance that is unique to each particular case. Experience shows that several instruments can work in different contexts. Environmental taxes have played an important role in some of the cases discussed, such as in China and Colombia; permit trading has had some success in pollution abatement in Chile (see Chapter 6) and in other countries’ fishing policies. Information provision and other soft instruments (e.g., voluntary agreements or two-tier regulation) have played important roles in Indonesia, Mexico, and India.

One might think that institutions must be in place before other policy instruments can function, and considerable logic and experience supports

such a notion. However, some issues are so pressing that they need immediate regulation, even before the institutions have been established. Building institutions requires time and money; building a functioning hazardous waste program takes at least 10–15 years, even when financial resources are relatively abundant and a law-abiding culture is present (Probst and Beierle 1999). For this reason, moderate (and earmarked) fees may be a suitable part of an early instrument package. Some issues are formally under the jurisdiction of national authorities, but in practice, multinational companies are keen on standardization. Their subsidiaries are sensitive to the pressures on foreign markets, and in such cases, information provision can be an extremely efficient tool. Even small fees and the obligation to provide correct information require some form of regulation and a capacity for monitoring and enforcement, so at least some institutional capacity must be established early. Instruments and institutions then develop in a process of continuous interaction.

It is impossible to generalize about distinct areas of the environment and natural resources, or about the groups of countries that are labeled “developing.” Some countries have an almost unbroken and cumulative historic development over many hundreds or even thousands of years that gives them special characteristics in such areas as rights and institutions that hinge on trust, culture, and norms. Other countries have a recent, conflictive, or chaotic history with great cultural diversity (e.g., created through colonialism). Some areas of environmental management are themselves dependent on complicated technological and ecological science (e.g., waste from chemical or nuclear facilities), whereas others are characterized by known technology but fraught with problems such as asymmetric information, as with the joint management of many natural resources.

Efficiency, however important, is not the sole criterion for policy selection. Considerable attention must be paid to distributional, informational, and political criteria as well as to the process of policy implementation itself. Particularly in developing countries, where monitoring and access to technology and credits are particularly difficult, it is crucial to consider policies that avoid antagonism and encourage cooperation and involvement.

Supplemental Reading

Blackman 2010	Seroa da Motta, Huber, and Ruitenbeek
Bluffstone and Larson 1997	1999
Cao et al. 2009	Shimshack and Ward 2005
Eskeland and Jimenez 1992	Tarhoaca and Zinnes 1996
Klarer, McNicholas, and Knaus 1999	Xu et al. 2010
Kathuria 2007	Vincent 2010
Markandya 1997	Zhang 1998

Notes

- 1 Chinese environmental authorities estimate that more than 70% of all pollution in China is industrial. However, in many middle-income countries such as Mexico and Brazil, motor vehicles and other sources create the majority of the emissions (World Bank 2000).
- 2 Finding descriptive labels for groups of countries is difficult. Words such as “rich,” “industrialized,” “OECD,” or “western” carry similar but imprecise connotations, especially when countries change categories. Poland, for instance, is poor by U.S. standards and was more so 20 years ago. However, the country even then was heavily industrialized—in fact, industrialization is the reason for many of Poland’s environmental problems. Although Poland could have been contrasted with OECD countries in the past, it has hardly been appropriate to do so since 1996, when the country joined the OECD (with Hungary, Korea, and Mexico). I refer to the generally wealthier group of “developing” countries (Poland, the Baltic States, the former Yugoslavian states, and so forth) as “central and eastern Europe” and the former Soviet Union (i.e., Russia, Belarus, and Ukraine) as the New Independent States. On the other hand, references to OECD countries typically mean the richer, more industrialized economies of western Europe, the United States, and Japan.
- 3 This section builds heavily on Afsah, Laplante, and Wheeler 1999, Wang and Wheeler 1996, and Wang and Wheeler 1999. It has greatly benefited from comments and information from Lee Travers, Hua Wang, David Wheeler, and Jian Xie.
- 4 *Visible pollution* is a measurement of particles and dust. The effect of invisible air pollutants (sulfur dioxide) and water (chemical oxygen demand) was much smaller.
- 5 Much of the information in this section is from the World Bank New Ideas in Pollution Regulation (NIPR) website (World Bank 2002). Thanks to David Wheeler of the World Bank, Thomas Black Arbelaez of MINAMBIENTE, Jorge Garcia and others of the Universidad de los Andes (Bogota, Colombia), and Allen Blackman of Resources for the Future for valuable comments and insights. As with many other schemes described in this book, social unrest is a constant threat to this kind of instrument (and to other efforts at building a sustainable society).
- 6 Thanks to Alan Blackman for comments on this section.
- 7 Wood accounts for more than 70% of energy requirements (Mulenga 1999). Although the township electrification program has connected many urban residents at a zero hook-up cost, many get disconnected because they cannot afford even the low rates.
- 8 Thanks to Shakeb Afsah for valuable comments and information for this section. Other sources for this section are World Bank 2000; Afsah, Laplante, and Makarim 1996; Pargal and Wheeler 1996; and Afsah and Vincent 1997.
- 9 This section is based on Kathuria and Sterner 2002. See also Murty, James, and Misra 1999; Kuik et al. 1997; Greenpeace 1999.
- 10 This landfill was the first site in India where a public consultation process was carried out as per World Bank/Ministry of Environment and Forests guidelines. It was due to a previous setback in 1996, when an attempt to set up a landfill near an adjoining village was stopped by stiff local resistance and public interest litigation was filed against the landfill.

PART VI

Policy Instruments for the Management of Natural Resources and Ecosystems

Management of natural resources and ecosystems—the “green issues”—is crucial for the world and economically vital in most developing countries. Ecosystem functions are both numerous and technologically and ecologically complex. The intricacy of an ecosystem is often mimicked by the social institutions designed to manage it. Risk management, uncertainty and information asymmetry, the risk of free riding, and distributional concerns must be addressed in the face of a partly stochastic outcome from nature. We do not pretend to cover all categories of natural resources in the next several chapters but have chosen a representative few that are ecologically significant, economically important, and illustrative of some theoretical issues of interest.

Due to the complexities related to natural resource management, dynamic optimization methods are needed. Dynamic optimization tools to derive optimality conditions are introduced in Chapter 22. The methods are illustrated by examples of the extraction of renewable resources, such as fisheries and forests, but they have also application to resource management in general.

The public sector has a natural role in providing (or at least monitoring) water supply because this vital “good” has many special characteristics: merit or public good, externalities, and “lumpiness” of productive investments. At the same time, managing water is much more than supply technology; pricing and tariff structure are crucial issues, as discussed in Chapter 23. Waste management

is seen as a “brown” environmental issue rather than a “green” natural resources issue. Nevertheless, waste management requires natural resources in the form of ecosystem capacity to recycle or assimilate waste (see Chapter 24).

Policy instruments for both large industrialized and small coastal fisheries are discussed in Chapter 25. Agriculture is addressed in Chapter 26, particularly regarding nonpoint-source pollution, property rights, incentives, and risk management. In Chapter 27, we discuss forestry, particularly the allocation of land between agriculture and forests. Understanding this allocation is essential to any discussion of policy instruments, which include title deeds, tenure security, common property management, logging fees, concession policies, stumpage fees, subsidies, tariffs, reduced emissions from deforestation and forest degradation (REDD), and other forms of international payments just to name a few. Finally, we explore ecosystem functions and their preservation through game reserves and ecotourism in Chapter 28.

Adapting Models to Ecosystems: Ecology, Time, and Space

Environmental and natural resources economics applies economic theory to the often complex world of ecosystems. In this chapter, we describe several sources of complication, starting with the biology of renewable resources such as fish and forests. The first models are for a single species, then the considerable complexities of real ecosystems are introduced. Next, we discuss dynamics—that is, development that takes time, which has both ecological and economic consequences. In the last section, we introduce some of the complications that arise as a result of geographical space. Finally, although many applications in this area concern natural resources, the theory applies equally to stock pollutants. Many chemicals are persistent and accumulate in the environment through various mechanisms. The capacity of an ecosystem to function despite an increasing stock of a certain pollutant may be limited, and all the economic models of how to “harvest” or “mine” a natural resource stock are equally applicable to the use of the assimilative capacity of the ecosystem with respect to stock pollutants (see Chapter 14).

A Simple Bioeconomic Model of a Fishery

One of the best-known bioeconomic models is that of a single-species fishery. This rudimentary model is useful primarily as a didactic and intellectual model; for empirical work it is less satisfactory because real fisheries are much more complex than the model reflects. Similar modeling may be used for other ecosystems that have some of the same general characteristics: left unmanaged, the ecosystem is only of limited use to humans; increased management can increase the yield or utility of the ecosystem; and too much management can destroy the ecosystem altogether.

Marrying ecological and economic modeling is difficult. In the simplest possible biological model, the net growth of a population is assumed to be a function of its size, which in itself gives exponential growth (i.e., more

BOX 22-1 Logistic Growth Curve

Growth in the stock size of a population can be modeled by the logistic growth curve. For small populations, growth is roughly exponential, because the number of offspring increases proportionally to the number of adults. However, some factor—space, food, energy, predation, or disease—always checks growth, and such factors are assumed to increase in importance as the population (N) approaches a carrying capacity (K). The result of these two tendencies is $(dN/dt)/N = g(K - N)/K$, where dN/dt is the population growth rate and g is the rate of growth when no restriction is imposed. When K is infinite, ordinary growth is exponential ($dN/dt = gN$), but when N is very close to K , growth is zero.

If it is assumed that fish can be sold at a constant price and that the net growth can be harvested without affecting the stock, then this growth equation can be interpreted as a curve showing the potential revenue for the resource (Figure 22-1). The growth in the fish stock is assumed to be the quantity that can be harvested without affecting the size of the stock. Therefore, every point of the curve in Figure 22-1 is an equilibrium point from the extinction equilibrium (O^*) to the natural population equilibrium (O).

reproductive adults lead to more offspring). (Net growth of biomass is the aggregate growth of individuals plus recruitment of new individuals minus natural mortality. In this section, growth and net growth are used interchangeably.) However, it is also assumed that the ecosystem has a capacity to support a certain population size, and as the population approaches this carrying capacity, mechanisms such as hunger, disease, and predation tend to reduce growth, giving the logistic growth curve (see Box 22-1). With appropriate assumptions, the logistic growth curve can be used to determine the potential revenue and costs of fishing for different long-run levels of fishing effort as well as stock size (as illustrated in Figure 22-1).

Increased effort generally gives higher harvest in the short run but reduces stocks (and harvest) in the long run. With no fishing effort, the natural equilibrium (O), yield is zero (otherwise it would not be a natural equilibrium), and stock (measured from the right origin) is at its maximum. At O , humanity does not receive any material benefits because no fish are caught (although there may be aesthetic, altruistic, existence, and other values). At the other end of the scale, a long-run maximum effort leads to stock depletion, or an extinction equilibrium (O^*), and thus yield is again zero. The model contains many other equilibriums; in fact, each point along the curve is a fishing equilibrium. However, some of them (such as O^*) may not be acceptable to future generations.

Noneconomists sometimes take for granted that the natural target of policy-making is maximum sustainable yield (MSY). The most important reason why MSY may not be a suitable target is that the costs of fishing typically increase with increasing scarcity (Figure 22-1).¹ Thus, a stock larger than at MSY would mean almost the same yield but at a considerably lower cost, giving a net profit

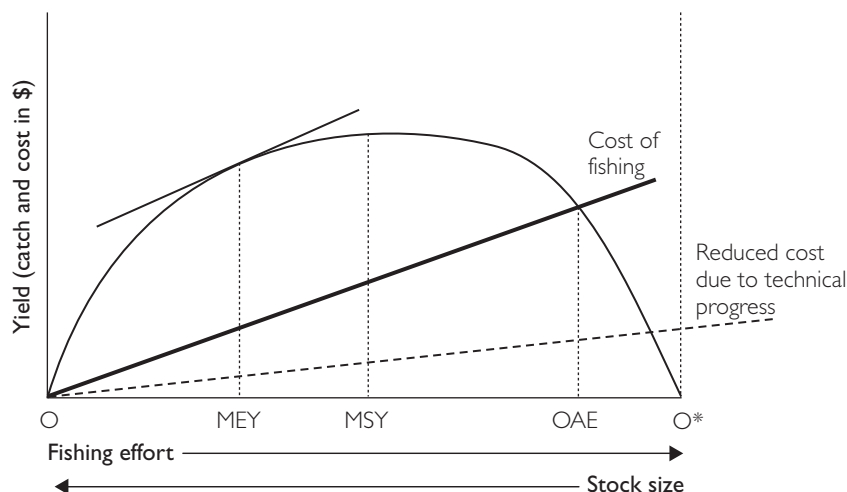


FIGURE 22-1 Simple Bioeconomic Model of Fishing

Notes: MEY = maximum economic yield; MSY = maximum sustainable yield; OAE = open-access equilibrium. Horizontal axis shows fishing effort and stock size measured from the extinction equilibrium (O^*). Vertical axis shows growth for each stock level; managing the fishery by harvesting the exact quantity that corresponds to this growth would be an *equilibrium strategy*. A simple cost of fishing function is also shown.

equal to the difference between harvest and costs (p^*); assuming a zero discount rate, this optimal policy is the maximum economic yield (MEY).

The logistic growth model shows that pure “hard-nosed” profit maximization can lead us to conclusions that ultimately are more, not less, conservationist than an intuitive model would be. It would be nice if the market worked this way; in fact, this is how the market would work if there were private rights to fish stock. In a private lake, the owner would manage the ecosystem according to this model if maximizing profits were the only goal. However, in most fishing grounds, the waters have no ownership, and the fish are too mobile to make property rights practical. Therefore, there is no scarcity rent, and each individual fisherman continues to fish up to the point at which marginal costs of fishing are covered by average harvest, open-access equilibrium.² This is how open access leads to over fishing, at which point there is no rent at all (total costs are equal to total revenue). The rents are completely dissipated, which translates to a loss of economic potential to the economy.

It is not reasonable to call the simple logistic growth curve for one species a biological—or an “ecological”—model, which, by definition, would have to account for interactions between species. However, combining it with an economic model helps us understand what fertile intellectual ground this kind of cross-breeding between disciplines can create. This combination was the foundation of the first fishery economics models used by Gordon (1954) and Schaefer (1954) and thus is often referred to as the Gordon–Schaefer

model. Modifications to this model abound. The dynamics are complicated (discussed in Scott 1955), but later applications of optimal control theory have greatly enhanced the analysis (Cohen 1987). For instance, the growth rate of the stock can be made to vary nonlinearly with stock size, as in $(dN/dt)/N = g(K - N)/K$. The growth rate can also be modeled in a more complex way, with discrete time, with harvest and growth in different seasons, and possibly with different age or size groups or cohorts. Some values of the parameter will imply that overfishing easily leads to extinction (if fish stocks fall below a critical threshold).

For some fish species, the relationship between recruitment (and thus growth) and stock may be much weaker, and in some models, recruitment (of “young” fish) is even modeled as exogenous or constant. For some species, the larvae float considerable distances in the ocean; thus, at least locally, recruitment is exogenous. Yet, some authors point to the strongly increasing relationship between egg production and size in some species—e.g., a 12.5-kg red snapper female can have more than 200 times as many eggs as a 1.1-kg red snapper (Pauly 1997). Therefore, recruitment might diminish drastically with fishing stock and be an argument for marine reserves where large fish would find a haven.

Similar reasoning underlies the metapopulation models (e.g., Brown and Roughgarden 1997), which explicitly model various adult populations of a species that are connected by a common larvae pool, such as lobster. The larvae pool may be pelagic (drift in the open sea) and thus spatially separated from the various subpopulations of adults in different sites that contribute larvae to and receive new recruits from it. In metapopulation models, economists typically see strongly increasing returns to scale, and the optimal policy turns out to be a strict specialization between sites that are suitable for harvesting (low costs because of distance to market and so forth) and sites that are best suited for reproduction. Taking the model literally, optimal harvest would entail harvesting all the adults in all sites except the one that is best suited to producing larvae. The conclusion for local fishing policies is thus different from the conclusions in the Gordon–Schaefer model: by harvesting all the adults, more harvest may be available in future periods because if there are excess larvae, then space, food, and suitable habitat are the only restricting parameters.

For demersal fish that live at the bottom of the ocean (e.g., flounder and haddock), metapopulation models indicate that recruitment is constant for a wide range of stock levels.

Fishing effort and the cost of fishing function may be quite complex. Some fish species swim together in large schools, so fishing costs do not rise as a simple function of increasing scarcity in stock. Much work has been done in the development of year–class models, which examine the effect of varying technology such as mesh size on the catches of different species and the age composition of the catch (which has effects on future growth and recruitment).

Recent research has grappled with the integration of some of the complexities of real ecosystems. A natural extension is to consider several

species of fish (some of which compete for food) and predation. The Lotka–Volterra biological model studies interactions such as predation by specifying dynamic equations for population: growth in the population of one species depends on its own numbers as well as the number of individuals in another population (Lotka 1932).³

Even strong believers in the market recognize that the fishing industry is one area in which policymaking is sorely needed. Although politicians have been active, the policies adopted often are the exact opposite of those needed. A prime example is subsidies to help fishermen purchase more equipment (e.g., boats, nets, and technology) when catches decline. The trouble is that more efficient equipment speeds up stock depletion and thus adds policy failure to market failure. The design of optimal policy is far from simple. Each fisherman creates externalities for other fishermen. Decreasing stocks increase the costs of fishing and lower future catches.

Clearly, fishing could be reduced through taxation; the understanding that fish are capital and should have a scarcity premium in situ also provides strong rationale for a tax. However, as illustrated in the congestion model (Chapter 2), implementing a tax leaves the immediate “beneficiaries” of the program worse off than without any policy. Tax solutions are not acceptable to fishermen because they take all the rent they help capture. Presumably for this reason, individual transferable quotas (ITQs) have become the main instrument of fishing policy. ITQs provide a scarcity signal but still leave the rent with the fishermen.

Policies must take into account technological advances (e.g., better ships, nets, logistics, and search equipment such as sonar). Such progress lowers the marginal cost of fishing (Figure 22-1), implying greater potential profits at the optimum but also bringing the open-access equilibrium dangerously close to the extinction equilibrium. This can lead to a negative attitude toward new technology; in fact, policies often have been designed to stop various technologies from being used. From a purely efficiency-oriented viewpoint, the optimal policy would be to use the new technology but reduce effort to a point corresponding to MEY. Common property resource management and ITQs, which appear to be the most promising instruments for providing appropriate trade-offs among efficiency, monitoring costs, and social acceptability in many cases, are discussed in Chapter 25.

A Simple Model of Forestry

Given the time period required for trees to grow from regeneration to maturity, the fundamental question of forest management is when a tree or a stand of trees should be harvested.⁴ The decision to harvest trees closely resembles the decision to sell a durable asset. However, in contrast to many assets, stands of trees often serve multiple purposes; besides timber,

and before harvest, most stands provide a range of nonmarket goods and services including a variety of aesthetic, ecological, and recreational goods and services. On the other hand, aging trees occupy land; if a forest owner wants to increase production, more land needs to be rented or purchased. Since land is scarce, the forest landowner must consider the opportunity cost of land when making the harvest decision.

The Faustmann model serves as a simple foundation for optimal harvesting when we consider only one rotation of trees—that is, seedlings are planted, the resulting stumpage is harvested at some point in the future, and the land is not replanted. In this model, landowners maximize the net present value (NPV) of land. All parameters are known and constant over time; individuals can lend and borrow unrestricted amounts at the prevailing interest rate r , and so are able to finance the liquidity constraints imposed by the use of land in forestry. The stumpage value per cubic meter of timber is p (the price of timber net of harvesting costs) and is the same for timber of all ages. $Q(t)$ is the growth function of the stand. Finally, c is stand regeneration cost, and $L(0, t)$ is the maximum NPV of bare land with stumpage harvested at age t .

The initial Faustmann optimal harvesting formulation (Faustmann 1849) is:

$$\begin{aligned} \text{Max NPV}_t &= L(0, t) = pQ(t)e^{-rt} - c + e^{-rt} [pQ(t)e^{-rt} - c] \\ &\quad + e^{-2rt} [pQ(t)e^{-rt} - c] + e^{-3rt} [pQ(t)e^{-rt} - c] + \dots \\ &= \frac{pQ(t)e^{-rt} - c}{1 - e^{-rt}} \end{aligned} \quad (22-1)$$

This formula assumes that the optimal rotation period will always be the same: the optimization problem at time 0 is identical to that at time T , after the first harvesting, and at time $2T$ and so on. It also assumes that the opportunity cost undertaken by using the land is given by the discounted value of future rotations. If harvest in a rotation is delayed, then the harvests in all the future rotations will also be delayed, reducing the present value of the future rotations. However, the opportunity costs of land may also come from nonforestry uses. Let W be the NPV of land from the most profitable nonforestry use and V be the maximum NPV of the land from either nonforestry or forestry use. Equation 22-2 can then be represented as:

$$\text{Max NPV}_t = pQ(t)e^{-rt} - c + e^{-rt}V \quad (22-2)$$

where $V = \max\{W, L(0, t)\}$. The optimal rotation period is then obtained from the first order condition with respect to t :

$$pQ'(t) = rpQ(t) + rV \quad (22-3)$$

This implies that a forest stand should be harvested when its value's rate of change with respect to time, $pQ'(t)$, equals the interest on the value of the timber, $rpQ(t)$, plus the interest on the forest land's value, rV .

If after the first rotation, nonforest uses are more valuable than forestry uses, W replaces V in Equation 22-3. Instead, if forestry is the best use, $L(0, t)$ replaces V and Equation 22-3 can be rewritten as:

$$pQ'(t) = \frac{r[pQ(t) - c]}{1 - e^{-rt}} \quad (22-4)$$

In equation 22-4, the numerator is the interest foregone by delaying harvest of the current rotation, while the denominator increases the interest foregone to reflect the delay on the harvest of all future rotations.

Comparative statics resulting from parameter changes in the Faustmann model are well known (see, e.g., Johansson and Löfgren 1985 and Brazee 2007). If the stumpage value increases, it is more profitable to harvest the timber faster and replant sooner, hence the optimal rotation period decreases. If replanting is becoming more costly, the optimal rotation period is lengthened such that replanting occurs less often. Formally, this is because higher replanting costs decrease the value of forest land, decreasing their interest costs; as a consequence, the rotation period lengthens and the value growth of the forest declines. Finally, an increase in the interest rate, *caeteris paribus*, will decrease the optimal rotation period since the costs associated with the interest rate on the value of the forest land are increased, outweighing the forest land's increase in value.

The assumptions behind this model are certainly strong; however, the results would not change significantly if we assume uncertainties about the future price of timber (the stumpage value) or the interest (Johansson and Löfgren 1985). Nevertheless, the violation of perfectly functioning land markets would certainly distort or invalidate the above results; insecure land tenure or impossibility to use land as collateral to finance short-term liquidity constraints will generate strong incentives to reduce forest planting or replanting, since the forest owner is less certain to benefit from the proceeds of the timber. The rotation period will thus decrease considerably, because a longer rotation period entails the risk of the agent not being able to exert his property right over the forest (Johansson and Löfgren 1985).

On the other hand, if externalities associated with forestry are not internalized and not taken into consideration by the private forest owners, the private and socially optimum rotation period will differ significantly. The positive externalities of forests provide society with benefits throughout the lifespan of the forest, whereas the forest owners gain benefits only at harvesting; as such, there will always be an under-provision of forestation (as is always the case with goods that generate positive externalities). However, the environmental benefits also depend on $Q(t)$; for example, the "carbon sequestration" function is more important in young forests, whereas old forests might be more

useful for habitat protection. To be able to assess how these environmental effects influence the rotation period, one needs to have precise information on the environmental effects of forests.

Another source of divergence between the two optimums is the difference in interest rates that the agents face: private interest rates are normally higher than the interest rates available to social planners, due to imperfect functioning of capital markets. This will lead private owners to choose lower rotation periods than are socially optimum, thus requiring that policymakers use different incentivizing mechanisms to motivate private owners to lengthen rotation periods.

Bioeconomics and the Management of Ecosystems

To capture the dynamics of two species is difficult enough, but true ecosystems often contain dozens of species, even if only the most vital ones at each trophic level are counted.⁵ Ecosystems are highly nonlinear complex adaptive systems (Levin 1998, 1999) that have many features bound to create market failures. This complexity leads to the existence of multiple domains of attraction (a counterpart to equilibriums in dynamic and chaotic systems) and to the possibility of sudden threshold changes in dynamics. For instance, long periods of drought in tropical drylands can lead to soil erosion followed by loss of tree cover and eventual desertification. Dry rangelands can alternate between two states of nature: with and without trees (Perrings and Walker 1997; Perrings and Stern 2000). The question is whether such changes are reversible within the time perspective of human decisionmakers.

Ecologists have shown that the resilience of ecosystems is an important factor in preventing irreversible change. Resilience depends on such things as the ecological memory of a system—its seed banks, migratory animals, and “patchiness” (i.e., the existence of heterogeneity in local conditions that may allow species to survive and recolonize a territory when conditions improve after a period of disturbance). The diversity of species at various levels, including the genetic level, is another mechanism that may enhance resilience in a way that is somewhat analogous to the way in which diversification reduces risk in a stock portfolio.

At the same time, the ecosystem is a source (from the human viewpoint) of joint production, where some of the products have characteristics of public goods or common pool resources. All these features make management difficult and argue in favor of adaptive management that relies heavily on the precautionary principle. Table 22-1 lists some of the ecosystem services that might be provided by a typical mangrove ecosystem, common in coastal areas in tropical countries.

Direct-use values accrue as a result of direct consumption. Indirect-use values might accrue for ecosystem services that indirectly give consumption

TABLE 22-1 Ecosystem Services from a Mangrove Ecosystem

<i>Use no.</i>	<i>Use value(s)</i>
1	Habitat, nutrients, and breeding of shrimp, crustaceans, and mollusks
2	Spawning and breeding of demersal (bottom of the ocean) fish species
3	Spawning and breeding of pelagic (open sea) fish species
4	Provision of fuelwood, construction wood, and so forth
5	Source of plants, herbs, and small game used locally
6	Production of salt and shrimp
7	Control of storms, floods, and erosion—extension of current land into the sea
8	Sequestration of carbon, nitrogen, phosphorus, and other nutrients
9	Protection of other marine and coastal ecosystems (e.g., coral reefs)
10	Sequestration of toxins
11	Protection of habitat for birds—amenity value of bird watching (i.e., tourism potential)
12	Option values related to future uses such as biotechnology and genetics
13	Existence (and bequest) value of ecosystem
14	Other exploitation

goods (e.g., rain). Nonuse values include intangible goods, such as scenic beauty. In Table 22-1, Uses 1–14 correspond roughly to direct-use values, indirect-use values, then nonuse values and one “alternative” use (discussed below). The sheer number of outputs might at first appear to make ecosystem management difficult; however, an ecosystem can be compared to a firm that produces many products. From a management viewpoint, quantifying the ecosystem characteristics and identifying the different beneficiaries (some of whom cannot be expected to pay for the services provided) for the many products is more difficult. The list therefore is also intended to differentiate among uses that are relevant for different categories of user, some of whom are poor.

Uses 1–3 (breeding) are related to fisheries in the broadest possible sense of the word, and the biology and economics of optimal yield could be analyzed using the models discussed in the fisheries section above. However, modeling is difficult for biological and economic reasons. Marine ecology is a multispecies system with complex and poorly understood dynamics; the economics are complicated by the social structures of the societies that benefit from the system’s resources. Typically, Uses 1 and 2 benefit many members of local communities by providing food, employment, and income. Use 3 typically benefits fishermen using large modern boats who originate in distant cities, even foreign countries.

Most marine and coastal bioresources, not owned by anyone, are therefore subject to overexploitation (through open access) and thus to rent dissipation as resources become scarce. With judicious ecosystem management, rents

could be restored (making the aggregate of users better off), but the level of coordination required to restrict overexploitation is difficult to achieve, even under ideal circumstances. Population growth and social mobility (sometimes coupled with famine, natural disasters, and war) can lead to large population influxes to coastal areas, with increasingly heterogeneous groups. Analysts of common property emphasize how exclusive control of a common pool resource and the continuous, coherent, and evolutionary social structure of groups dependent on a common pool resource are important attributes for the success of such populations (see Chapter 8). Several distinct groups enjoying or using different resources provided jointly by the same ecosystem adds a layer of complexity.

The chance of these user groups interacting is small. Japanese fishing vessels trawling for fish in the open seas of the Gulf of Thailand benefit from the “spawning services” of the mangrove forest, but they probably do not interact with local communities to help them protect the mangroves. In fact, anecdotal evidence indicates that many Japanese fishermen invest in shrimp farming—an industry that creates large ponds by clearing mangroves, thereby eliminating the spawning grounds on which their open-sea fishing depends.

Uses 4 and 5 (provision of fuelwood and other plant products) are typically straightforward goods. Some of these goods may be difficult to value because they are used only locally and lack market prices. Data must be collected on quantities and other use characteristics. If the products are used by poor local people and are necessary proteins, medicines, or other products similarly essential to human life, their loss may be considered a greater loss of welfare than would be suggested by a simple valuation of close commercial substitutes. Another complicating factor is that these goods could be collected by social groups other than those who use the aquatic resources. Use 6 (salt and shrimp production) may be categorized with Uses 4 and 5 if done by small-scale operations.

Uses 7–9 (protection against storms and erosion, sequestration of some nutrients, and the associated protection of other marine ecosystems) are ecosystem services par excellence: they have true public good or common pool resource characteristics that benefit all local inhabitants. The users may or may not be the same as those who benefit from Uses 1–6. In some areas, such as off the coast of Kenya and Tanzania, the mangroves along river estuaries and coasts may protect offshore coral islands with a separate fishing population—again raising difficult issues of collaboration. The trapping of nutrients may have external benefits, such as protecting coral reefs, which are sensitive to large amounts of nutrients. Compensation in this case might seem to be a remote possibility; the fishermen on the coral reefs cannot easily compensate or interact with those who cut down the mangroves. However, the global community is starting to create mechanisms to compensate for additional positive global externalities associated with local projects.

Use 10 (sequestering toxins to prevent damage farther downstream) is applicable only to mangrove ecosystems near industries or subject to pesticide

runoff from agriculture or effluents from cities. This use precludes many of the other uses, but it may be a cost-effective alternative to sewage treatment. For Uses 9 and 11 (protection of the coastal areas and amenity values for tourists), local decisionmakers are less concerned with consumer surplus than with revenue generation. However, income from tourism generally benefits players other than the local fishermen.

Uses 12 and 13 are nonuse values: option value (Use 12) relates to the value of keeping an “option” open (e.g., to use a resource in a new way in the future), whereas existence value (Use 13) is an intrinsic value that one might experience from just knowing that something exists, even if one does not expect to see it (e.g., a whale). Nonuse values can be local or global and are generally difficult to capture in practice. (One exception is when biomedical companies and local communities enter contracts to conduct joint research seeking species that can help produce new drugs; with such a contract, these benefits can be judged in the same commercial way as tourist income.)

Some uses can be combined, whereas others are exclusive. A debate or policy issue is often triggered by a proposal from an entrepreneur who is interested in large-scale industry, a port, a refinery, salt production, aquaculture (shrimp production), or conversion to agriculture. Such projects appear to promise large revenues and massive job opportunities. Examples abound, from the Rufiji Delta in Tanzania—where the African Fish Company was initially given a lease of 10,000 hectares but later stopped in response to protests against large-scale destruction of the mangroves—to Florida, where the expansion of Miami, Orlando, and other cities has pushed the Everglades to the verge of collapse through demands for land, water, and flood control. In the Florida Everglades, 90% of the wading bird population has become extinct, and \$8 billion will be spent over the next 20 years to restore natural water flow and thereby vegetation and ecosystem functions.

Considerable uncertainty surrounds ecosystem function and the valuations of the many services that ecosystems provide. Because there are no market values to observe, it is crucial to model the systems to obtain some good estimates. Necessity dictates the use of drastic simplifications, but the appropriate simplifications must be chosen for the type of problem at hand. For a mangrove swamp, the variables might be the area covered (A), the biomass (M), fish stock (N), and nutrients in the soil (S). By denoting the harvest of fish and wood (H_N and H_M , respectively), the growth rates of fish and wood can be modeled by using equations that are reminiscent of those in the Gordon–Schaefer model. Denoting unimpeded growth rates (g) and the carrying capacity (K) with appropriate subscripts, the equation for fish is:

$$\frac{dN}{dt} = g_N(A, S)N \left[K_N(A, S) - N \right] - H_N \quad (22-5)$$

The equation for wood is analogous.

Equation 22-5 does not account for species, age composition, and many details of recruitment and mortality, but it may be a reasonable approximation for focusing on the dynamics of the interaction among mangrove area, fish stock, and nutrients. For other types of ecosystems, it may be reasonable to concentrate on the variables whose dynamics are applicable in a shorter time span and abstract the dynamics of “slow” variables such as forest area and biomass (see Levin and Pacala 1997 for theories of model simplification). Still, estimating dynamic equations such as Equation 22-5 is a daunting task. Waiting for time series data is hardly attractive, and using cross-sections may be unreliable because ecosystems are so idiosyncratic; the relationships governing one swamp may be different from those in a superficially similar one (Cole, Lovett, and Findlay 1991).

Still, thresholds and surprises may be extremely hard to foresee in modeling an ecosystem. Another neglected feature of ecosystems is their typically strong thresholds: ecosystems may “flip” quickly from one state to another or exhibit strong irreversibilities. This feature is illustrated in Figure 22-2. The curve has a shape similar to the one for single-species fisheries in Figure 22-1, except that one section with intense use is extremely steep (or nonexistent), reflecting a threshold or irreversibility. Eutrophication in a shallow lake may have this characteristic. A small inflow of phosphorus increases algae growth, reducing light and oxygen at the lake bottom. If plants on the bottom are killed, then the sediment may be destabilized, leading to release of much more phosphorus in an almost irreversible spiral. In such cases, ordinary optimization (as represented by adaptation to the utility curve shown) can easily drive the system from a desirable state (point A) to an undesirable suboptimum state (point B), from which recovery can be extremely expensive. With true irreversibility, recovery might be impossible, but truly irreversible cases (luckily) appear to be relatively rare.

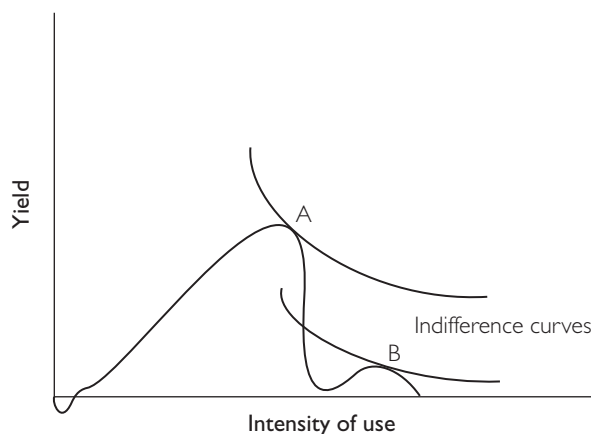


FIGURE 22-2 Managing an Ecosystem with Threshold Effects

Oversimplification does have its pitfalls and is the basis for the so-called precautionary principle, which suggests that if the optimum is not known, then one should err on the side of caution. This suggestion sounds straightforward but can be too stifling for development. A reasonable strategy may be a form of careful adaptive management. By allowing small changes and continuously monitoring the ecosystem, not only are the interests of various stakeholders catered to, but valuable information gained allows modification of the management strategy itself. Thus, if shrimp farming is being proposed, it would seem wise to start with a small-scale activity rather than a large-scale project like the Rufiji Delta.

To recap: The range of challenges is wide. A diversity of services are produced jointly by the same ecosystem but used by different users. The ecosystem is complex, dynamic, and often poorly understood. The definition of property rights is incomplete; sometimes, conflict arises from claims between local communities (which exercise traditional and culturally based rights) and outside society (represented by new immigrants, new classes of entrepreneur, the state, or foreign users). The high costs of enforcement and the lack of clear property rights may create situations of almost open access at the local level. Even if local users can collaborate, it is difficult for them to handle external effects that have repercussions on other groups. The agents using the various resources or ecosystem functions have little experience in negotiating with each other. Compounding the problem for the central policymaker are tricky issues of market structure, uncertainty, aggregation, and discounting.

Although these problems may sound insurmountable, they must be resolved. About two thirds of the world's current population lives in coastal areas; this proportion is estimated to increase to three quarters (more than 6 billion) by 2025 (Lindén and Granlund 1998). In addition, fish is the main source of protein for the majority of poor people. The first step in analyzing these problems must be to isolate the most crucial conflicts of interest. When overfishing is the main problem and fishing effort needs to be restricted, severe conflicts may arise concerning who is to be allowed to continue fishing. Also, defining fishing rights or quotas may be necessary policy; when the most crucial conflict of interest is between local fishermen and the expansion of aquaculture or industry, types of regulation such as pollution control or zoning may be more appropriate. Most experience to date suggests that integrated coastal zone management (ICZM) is a promising policy instrument for this type of situation. ICZM is a constant learning process at the scientific and institutional levels that entails bringing together all concerned stakeholders, carefully monitoring the local fisheries, conducting research, promoting local learning and capacity building, strengthening local institutions, providing feedback to the stakeholders, and, ultimately, designing new improved programs. Local management, incentive compatibility, involvement, property rights, and public information are key elements of ICZM.

Management in an Intertemporal Setting

Many problems related to natural resources or the environment involve stocks as well as flows; thus, the time dimension is an essential component.⁶ Whenever decisions in the present affect the future, a dynamic perspective is important.

Early applications of dynamic methods in environmental and resource economics analyzed typical renewable examples—fish, livestock, or timber—and typical exhaustible examples—fuel and nonfuel minerals. Nevertheless, more recently, dynamic methods have been applied to pollutant accumulation (Goetz and Zilberman 2000), carbon storage (Sohngen and Mendelsohn 2003; Sohngen et al. 2009), endangered species (Brown and Layton 2001), soil erosion (Ekbom et al. 2009), natural environments (Barbier 2009), even resistance to antibiotics (Laxminarayan and Brown 2001).

The distinction between exhaustible and renewable aspects of the natural world is flexible. For example, even renewable aspects such as biological growth may be exhausted when the rate of use is high enough or the rate of growth low enough (e.g., Clark 1976, Section 2.8). Exhaustible resources may be renewed through discovery (e.g., Fisher 1981, Section 2.7). Their benefits may be extended, or even replaced, through technological innovation (e.g., Dasgupta and Heal 1980, Chapter 7). Finite capacities of air, water, or land environments to absorb pollution may be changed through meteorological, hydrological, or terrestrial processes (e.g., Goetz and Zilberman 2000). Populations of biological species may increase through reproduction, or be changed through immigration, emigration, or predation (e.g., Sanchirico and Wilen 1999).

Regardless of the modifications made for a specific study, a typical dynamic problem includes consequences for both present and future, and at least one constraint that links present and future. Consequently the results obtained must include those that hold *within* a period, and those that hold *between* periods. The amount that evolves over time is often referred to as the stock, and its rate of change is often referred to as the flow. The flow may be influenced by economic decisions such as extraction, harvest, or policies.

Lagrangian or Hamiltonian functions are the two most common functions that include the necessary constraints in dynamic problems. Lagrangian methods are commonly applied to discrete time problems, and Hamiltonian methods are commonly applied to continuous time problems, but either method can readily be applied to discrete or continuous problems. Discrete time specifications have the advantage that they can be adapted to numerical simulations using readily available software packages (e.g., Conrad 2010).

To see what these approaches have in common, consider a manager who must decide how much, if any, of a finite stock of oil should be extracted over a time period. It is often easiest to begin with a discrete time Lagrangian specification, because Lagrangians usually include the entire planning horizon and a constraint for every period. In contrast, a Hamiltonian typically includes one period and employs auxiliary equations such as costate equations and

transversality conditions to extend the optimal solution throughout the time horizon (e.g., Dorfman 1969). Write the initial stock of oil as Q_0 , the stock at the beginning of time t as Q_t , and the amount extracted between t and $t + 1$ as q_t . This means q_t is equal to $Q_t - Q_{t+1}$.

Oil is extracted at a total cost of $c(q_t, Q_t)$. Marginal cost of extraction is $\partial c / \partial q_t$. The effect of stock on cost is $\partial c / \partial Q_t$ and is referred to as a stock effect. Because it is often cheaper to extract oil when it is more abundant, it is usually presumed that $\partial c / \partial Q_t < 0$.

It is common to use a partial equilibrium approach, in which the value of extracted oil is measured by its price, $p(q_t)$. Net benefits in period t (consumer plus producer surplus) may be written as $\int_{0, q_t} p(\xi) d\xi$ minus $c(q_t, Q_t)$. An economic welfare objective is often used to examine socially efficient extraction. Some authors compare privately efficient and socially efficient extraction to evaluate whether policies are needed to reconcile the rates (e.g., Fisher 1981, Chapter 2).

With perfect property rights, no market power, no public goods, no external costs, and equal private and social rates of time preference, these rates are the same and there is no need for policies to reconcile them. However, there is no particular reason to expect these conditions to be met in real world applications. Consequently, policies are usually needed to help align private environmental and resource decisions with social goals.

A Lagrangian specification for this problem may include the net benefits and constraints for each period t , where t ranges from 0 to T . Present values are often used, but current values can yield equivalent results. The objective is to choose q_t to maximize the present value of net benefits,

$$\sum_{0,T} (\int_{0, q_t} p(\xi) d\xi - c(q_t, Q_t)) / (1 + r)^t \quad (22-6)$$

subject to the constraint that $q_t = Q_t - Q_{t+1}$ for each period from 0 to $T - 1$. Together with the objective, the Lagrangian function for this problem includes:

$$L = \sum_{0,T} (\int_{0, q_t} p(\xi) d\xi - c(q_t, Q_t)) / (1 + r)^t + \sum_{0,T-1} \lambda_t (Q_t - Q_{t+1} - q_t) \quad (22-7)$$

The multiplier λ_t is often called the shadow value of the constraint that $Q_t - Q_{t+1}$ equals q_t . This value can be interpreted as the marginal value of one more unit of oil at time t . Because we have written the objective as present value, λ_t measures the marginal present value of a unit of oil in the ground (unextracted) at time t .

The optimal stock and flow of oil in each period are obtained from the first order conditions required for Q_t and q_t .

$$\partial L / \partial q_t = (p(q_t) - \partial c / \partial q_t) / (1 + r)^t - \lambda_t \quad (22-8)$$

$$\partial L / \partial Q_t = -\partial c / \partial Q_t / (1 + r)^t + \lambda_t - \lambda_{t-1} \quad (22-9)$$

Provided that it is optimal to extract some amount of oil in period t , optimal q_t will be positive and optimal $\partial L/\partial q_t$ will be zero. With some rearranging, this requires

$$p(q_t) - \partial c/\partial q_t = \lambda_t(1 + r)^t. \quad (22-10)$$

That is, within each period t , the marginal benefit from extracting the resource, $p(q_t)$, should equal the marginal cost of extracting it, $\partial c/\partial q_t$, plus the marginal opportunity cost of having less of it, $\lambda_t(1 + r)^t$.

Conditions for optimal Q_t describe the requirements that hold between periods. Provided that it is optimal to have some oil left in the ground, optimal Q_t will be positive and optimal $\partial L/\partial Q_t$ will be zero. Multiplying $\partial L/\partial Q_t$ by $(1 + r)^t$ and rearranging yields Equation 22-10 becomes:

$$-\partial c/\partial Q_t + \lambda_t(1 + r)^t - \lambda_{t-1}(1 + r)^{t-1} = r\lambda_{t-1}(1 + r)^{t-1} \quad (22-11)$$

This is easier to interpret if we use μ_t to represent the marginal current value of the resource in period t , $\lambda_t(1 + r)^t$,

$$-\partial c/\partial Q_t + (\mu_t - \mu_{t-1}) = r\mu_{t-1} \quad (22-12)$$

The marginal benefits of conservation (leaving the oil in the ground) from $t - 1$ to t include the effect of stock reducing extraction costs, $-\partial c/\partial Q_t$, and gains in the resource value, $\mu_t - \mu_{t-1}$. These marginal benefits to conservation must equal the marginal cost of conservation, $r\mu_{t-1}$. This marginal cost is the opportunity cost of not extracting oil worth μ_{t-1} in period $t - 1$, and investing the proceeds at the rate r (the opportunity cost of capital invested in the resource) for one period.

Many authors express the gains in the resource value, $\mu_t - \mu_{t-1}$ as a rate. In the absence of stock effects, $(\mu_t - \mu_{t-1})/\mu_{t-1} = r$, which means when the resource is optimally extracted, its marginal (current) value will grow at the rate of time preference, r .

When monopoly revenue is used as an objective, marginal revenue, $p(q_t) + p(q_t)q_t$, must grow at this rate. This extraction is initially slower than socially efficient extraction, because the monopolist optimally conserves the resource to take advantage of market power and keep prices higher. The value of the resource can be obtained by substituting the optimum series of q_t back into the objective function (Hotelling 1931).

Now consider a continuous time Hamiltonian specification for this optimal extraction problem. The Hamiltonian function is for one period within the objective function, the flow constraint describes how the stock changes, a costate equation describes how the optimal shadow price changes, and transversality conditions describe how the shadow prices should be at the end of the planning horizon.⁷

In continuous time, the rate of change in the resource stock is $dQ_t/dt = -q_t$, and the current-value Hamiltonian for this problem may be written

$$H = \left(\int_{0,q_t} p(\xi) d\xi - c(q_t, Q_t) \right) + \mu_t(-q_t) \quad (22-13)$$

In a Hamiltonian, the extraction rate q_t may be referred to as the control variable, the stock Q_t may be referred to as the state variable, and the multiplier μ_t may be referred to as the costate variable.

As in the Lagrangian specification, the variable μ_t may be interpreted as the shadow value of an additional unit of the resource at time t . Transversality conditions describe requirements for μ_t at time T , but these are of less importance here (see Dorfman 1969 Section III on boundary conditions for further discussion).

The optimal extraction is found by conditions on q_t ,

$$\partial H / \partial q_t = (p(q_t) - \partial c / \partial q_t) - \mu_t \quad (22-14)$$

Provided that it is optimal to extract some amount of oil in period t , optimal q_t will be positive and optimal $\partial H / \partial q_t$ will be zero. With some rearranging, this requires

$$p(q_t) - \partial c / \partial q_t = \mu_t \quad (22-15)$$

which means that the marginal current value of the resource is its value in the market, $p(q_t)$, minus the marginal cost of extracting it, $\partial c / \partial q_t$. This condition holds within each period where the optimal extraction is positive and less than the maximum extraction rate.

For this current-value Hamiltonian, the costate equation describes the optimal path for the costate variable,

$$d\mu_t/dt = r\mu_t - \partial H / \partial Q_t \quad (22-16)$$

For comparison with the Lagrangian results, it is convenient to rewrite this condition in terms of incentives to conserve or develop (extract) the stock,

$$d\mu_t/dt - \partial c / \partial Q_t = r\mu_t \quad (22-17)$$

Marginal benefits of conserving the stock include capital gains, $d\mu_t/dt$ and the stock effect, $-\partial c / \partial Q_t$. Marginal opportunity costs of conserving the stock include $r\mu_t$. This condition can also be expressed as a rate by solving for r . For comparison purposes, consider the result we would have with no stock effects. Then $(d\mu_t/dt)/\mu_t = r$, which requires the optimal current value of the stock to grow at the rate of time preference, r . Therefore, optimal control and Hamiltonians allow us to analyze trajectories in extraction and stock of a resource over time.

Dynamic Problems in Environmental and Resource Economics: Fisheries and Forests

For renewable resources, the interaction between harvest and biological rates of reproduction or growth affects the results of economic optimization over time considerably. Still, the effect of introducing a discount rate is analogous to that in the Hotelling model for a finite stock resource. For a simple bioeconomic model of fishing, for instance, the optimum typically would not be the same point as MEY in Figure 22-1. Instead, the optimum would be moved to the right, to a somewhat higher level of fishing effort and lower stock size. This shift would be accompanied by a short-term increase in catch that would lower returns in the future; the proceeds of this drawing down the natural stock are invested to yield a higher rate of return elsewhere in the economy. In this model, the result may be a steady state (with constant catch each year), even though resource rents rise at the rate of discount.

For forests, policy must account for several rotations (i.e., cycles of planting, growth, and harvest) on the same acreage. By including the expected discounted value of future forest rotations, Faustmann correctly showed that forest rotations would be shorter with than without discounting (i.e., advancing the harvest not only brings in cash faster from current timber but also shortens the time period by which future harvest will have to be discounted).

In terms of the optimization problem, after the forest has been planted, at the end of each Faustmann rotation, harvest revenues of $pQ(t)$ are obtained, and a new forest must be planted at a cost of c . This is a repeating series of net payments of $[pQ(t) - c]$ that begins after t years. When (continuously) discounted at the rate r , and subtracting the initial cost of the first rotation, the present value of the periodic series that begins in year t and repeats every t years is:

$$J(t) = \{[pQ(t) - c]/[e^{rt} - 1]\} - c \quad (22-18)$$

which can be solved for optimum t by applying the quotient rule to $J(t)$. An interior solution for t is given by $dJ(t)/dt = 0$, and after some simplifications can be represented as:

$$pQ(t)/[pQ(t) - c] = r/[1 - e^{-rt}] \quad (22-19)$$

This expression requires the growth in value in the forest to be higher than the interest rate, since $1 - e^{-rt}$ will be less than one. This adjustment is a manifestation of the opportunity cost of delaying future stands on this land.

Equivalently, this problem can be represented in terms of the marginal benefit and marginal (opportunity) cost of extending the rotation (t) by one period. One commonly used form for the objective function is

$$J(t) = \{[pQ(t)e^{-rt} - c]/[1 - e^{-rt}]\} \quad (22-20)$$

From where we get the condition:

$$pQ(t^*) = rJ(t^*) + rpQ(t^*) \quad (22-21)$$

In other words, the marginal benefit of extending the rotation, $pQ(t^*)$, must equal the marginal opportunity costs of the land, $rJ(t^*)$, and the capital stock (the timber), $rpQ(t^*)$. When an existing forest is younger than t^* , it is optimal to grow to age t^* . When an existing forest is older than t^* it is optimal to immediately harvest. In both cases, subsequent rotations are grown to age t^* .

When the standing forest provides annual benefits of $a(t)$, these can also be considered benefits from the forest stock. The function $a(t)$ can be considered an amenity benefit function, which may or may not include benefits valued in markets. Considering these benefits provides a vital economic reason to preserve natural forest environments that is not present in simpler problems.

We can write the combined timber and amenity benefits from this rotation as

$$J(t) = \{[pQ(t) e^{-rt} + (\int_{0,t} a(t) e^{-rt}) - c]/[1 - e^{-rt}]\} \quad (22-22)$$

which can be solved for optimum t , leading to the interior solution:

$$pQ(t) + a(t) = rJ(t) + rpQ(t) \quad (22-23)$$

In this case, the marginal benefit of extending the rotation, $pQ(t)$, plus one year's additional amenity benefit, $a(t)$, must equal the marginal opportunity costs of the land, $rJ(t)$, and the capital stock (the timber), $rpQ(t)$. This result has very important implications for preservation of natural forests. In the Faustmann rotation, when an existing forest is older than t^* it is optimal to immediately harvest. This is because in the Faustmann rotation, returns to the time input are eventually presumed to diminish. In contrast, when amenity benefits are present and do not decrease with time, then it is possible that $pQ(t) + a(t) > rJ(t) + rpQ(t)$, for all time. This means that the optimum rotation for this forest is of infinite length. In other words, this forest should be dedicated to permanent provision of amenity benefits.

Spatial Heterogeneity and Land Use

One essential aspect of ecosystems is spatial heterogeneity or variation.⁸ To build good models for ecological economics, economists must account for spatial dimensions—something they are somewhat unaccustomed to doing. Metapopulation models for a marine ecosystem (in which groups of adults and larvae are spatially separated) illustrate how spatial dimensions lead to conclusions different from those produced by the simple Gordon–Schaefer

fishery models. Particularly in the areas of biodiversity preservation, parks management, and the protection of genetic diversity, it is crucial to consider the patchiness of nature.

A model of spatial heterogeneity can be used to analyze land use and deforestation. To understand changes in land use in general and deforestation in particular, and to suggest policy instruments, it is necessary to first understand what a forest is. It is not just a collection of trees but a whole system that includes the soil and its flow of water and nutrients to animals (including insects and microorganisms), which typically play important ecological roles in forest development. In this section, we distinguish natural forests—defined by high biodiversity and low human density and interference—from plantation forests, which are more or less simple monocultures. Much confusion can be avoided by carefully distinguishing these two categories.

Consider a stylized pattern of forest development. The first phase of a new settlement involves rural development and deforestation. In later stages, forest products become increasingly scarce, and prices increase. Eventually, forest investment could limit or even reverse deforestation. This pattern follows the concepts of economic geography first proposed by von Thünen in 1826 (see Samuelson 1983 for a review). The basic elements of this pattern are illustrated in Figure 22-3. Similar analyses can be applied to the allocation between more intensive and less intensive land uses, such as ranching, agriculture, and even fishing versus aquaculture.

The assumptions made concerning the determinants of agricultural development are necessarily simplified for the scope of this book. Agriculture

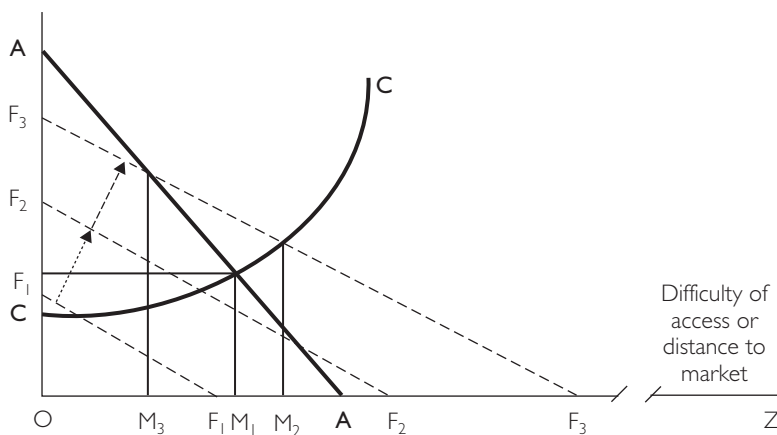


FIGURE 22-3 Economic Determination of the Margins of Forestry vis-à-vis Agriculture in Different Stages of Development

Notes: AA = value of agricultural land; FF (with subscripts) = a series of curves that depict the increasing value of forestland; CC = cost of protecting private property rights; M_1 , M_2 , and M_3 = margins of agriculture or forestry.

is a particularly risky and stochastic enterprise, and expensive monitoring creates special problems for the labor and insurance markets (discussed in Chapters 2, 10, and 26). Nevertheless, Figure 22-3 illustrates a simple linear landscape of agriculture and forests. The origin (O) identifies the location of a new settlement and market. Land extends out into virgin forest at some distant point Z. The value of land around the settlement (A–A) is a function of the net agricultural prices at the farm gate—which are greatest when the farm is located near the market, partly because of transport costs and partly because the settlement is located in an area where the land is most productive. The value of agricultural land declines with increased distance to the market until it eventually reaches zero at Point A.

A natural cover of mature forest occupies the landscape at the time of initial settlement. The forest closest to settlement may have a negative value because it interferes with agriculture. Settlers remove the trees wherever the agricultural value of converted forest plus the value of harvesting the trees (e.g., for fuel or construction) exceeds removal costs.⁹ They may leave some trees standing and plant around them. Forest value is depicted as a series of curves F–F that shift outward over time. Initially, the values for forest land are much lower than for agricultural land (because food is more essential to survival than trees are) and also somewhat “flatter,” because transport costs are assumed to be less important (because timber is less perishable than food).

Landowners must pay (C–C) to claim and defend property rights. These costs are assumed to increase as the level of public presence and thus effective control declines farther from the market center. (The cost of protection may vary depending on whether the land is used for forestry or agriculture; this aspect is simplified for the purposes of this chapter.) In the first phase, the extent of agricultural land is decided by the intersection of the value of agricultural land and the cost of property rights. Thus, farming will extend from O to M_1 . Farther out (from M_1 to Z), the net value of the forest timber is assumed to be negative, and the natural forest will be managed as an open-access resource to be exploited for its native fruits and nuts, for forage and fodder.

Notice that in this first phase, deforestation is caused by the demand for agricultural land. The market price of forest products must cover the costs of production. However, growing the forest entails no costs, because the forest is already mature. The only costs of forest production are harvest and delivery costs. In this kind of forest, such policy instruments as forest taxes or subsidies (such as free seedlings) will have little or no effect because deforestation is caused by the need for agricultural land and not by the profitability of forestry.

As incomes rise, technology and tastes change, and infrastructural improvements reduce transport and other costs, the forest value function may rise to F_2 – F_2 . This increase would imply deforestation from M_1 to F_2 . In this second phase, the agricultural frontier would be at M_1 and a degraded forest area from M_1 to F_2 . This area is still an open-access natural forest, with no costs for management or property rights, because values are lower

than the cost of protecting property rights. This area would be logged at least once and then perhaps abandoned unless it were profitable for agriculture. Deforestation in this situation would be explained by timber prices and other parameters relevant to forestry. Agricultural land conversion would no longer be a source of deforestation, because farming would take place primarily in the already deforested area, M_1 – F_2 . In this case, forestry policy instruments that affect the cost of harvesting would limit deforestation, but agricultural policies and free seedlings (which only affect plantation costs) would not.

It is possible that wages would eventually rise, making agriculture relatively less profitable than forestry. However, technological progress counteracts the effect of rising wages, and experience in the United States and Scandinavia indicates that rising wages favor forestry in the long run because it requires less labor. As timber scarcity increases, the frontier of economic forest activity may extend to F_3 – F_3 . Forestry would become more profitable than agriculture, even on some prime land that was formerly farmed.¹⁰ Agricultural land would be limited to O – M_3 , whereas new managed plantation forests would cover M_3 – M_2 . The degraded open-access region would extend from M_2 to F_3 , and timber would have two distinct sources: the plantation forests, protected by secure property rights, and the natural forest at the frontier. (If property rights are secure, then according to the Faustmann formula, forestry management on plantations will be optimized over time [Johansson and Löfgren 1985].) For the usual reasons, the marginal cost of growing, harvesting, and delivering products from the managed forest is expected to be equalized to the marginal cost of harvesting and delivering from the more distant natural forest. Different agricultural and silvicultural policies would have different effects on the various margins or regions (e.g., the provision of free seedlings might increase forest growth in a plantation forest but not in a natural forest).

Perhaps the most interesting characteristic of forests for policy analysis is their economic margins. The first two stages of forest development have only one economic margin (at points M_1 for the first stage and F_2 for the second), whereas the third stage has the truly unusual feature of three margins: intensive and extensive managed forest margins at M_3 and M_2 , respectively, and the margin of harvests from the mature natural forest at F_3 . The margins of the third stage in particular create some unusual policy implications. (The importance of these issues for forest policy is discussed in Chapter 27).

Many countries have regions that can be described by one of the three possible stages of forest development discussed in this chapter. However, some might require different models, because forests and social settings are heterogeneous. Some forests have valuable tree species for which the timber value is far greater than any agricultural value the land might have. In such instances, deforestation might be of primary importance, and agriculture or ranching would be a secondary activity. Some countries have open access to resources, whereas others enforce private or state property rights to some extent. Some

have large populations that hunger for productive land, whereas others have small populations and less scarcity of land.

Most forest products are either bulky or perishable and cannot be transported cheaply. Therefore, their primary markets are often geographically contained, and primary processing facilities are located near the source. Large regions of mature natural forest still exist in many parts of the world, but other parts of the world support managed forests. In some areas, such as Scandinavia and the northern United States, managed forests have been growing consistently for several decades, implying—to the surprise of some—that this (spontaneous) reforestation is a significant source of carbon sequestration in some of the most industrialized countries. However, even the southern United States (often thought to be the most dynamic region of the world for commercial timber production) still harvests more than half of its timber from unmanaged stands.

Supplemental Reading

Barbier 2009	Laxminarayan and Brown 2001
Binswanger and Deininger 1997	Munro and Scott 1985
Brazee 2007	Sanchirico and Springborn
Clark 1990	2011
Conrad 2010	Stiglitz 1999
Eggert 1998	Tschirhart 2009

Notes

- 1 Other reasons why MSY may not be a suitable policymaking target are uncertainty and discounting.
- 2 *Scarcity rent* is the rent that accrues to the owner of a natural resource just because it is scarce. Throughout most of history, land has been scarce and thus has had value. Scarcity did not use to be a problem for ecosystem services such as clean air and water; however, as scarcity becomes more widespread, it must be internalized.
- 3 Some models analyze the difference between harvest rates and harvest quotas in multispecies models (e.g., Conrad and Adu-Asamoah 1986; Flaaten 1988; Azar, Holmberg, and Lindgren 1995). The “dove–hawk” model, also relevant, uses game theory to study the evolutionary effects of having two different strategies for two groups within a population. The doves are peaceful feeders that avoid conflicts, whereas the hawks always fight food competitors. Hawk aggressivity brings advantages but also costs, because of the risk of injury or death. This simple model creates interesting dynamic patterns of interaction with free riding and explains many biological (and economic) phenomena (Smith 1982).
- 4 This section is based on Brazee 2007.
- 5 This section draws on Arrow et al. 2000.

- 6 This section was written in collaboration with Peter Parks.
- 7 Dorfman (1969), Clark (1976), Seierstad and Sydsæter (1987), Kamien and Schwartz (1991), and Chiang (1992), provide considerably more detail on these topics.
- 8 This section was written in collaboration with Bill Hyde.
- 9 In other cases, some exceptional hardwoods may have timber value. The focus here is on the settlers, who are in fact the cause of deforestation. In addition, “untouched” forests may be home to indigenous peoples who typically survive by hunting and gathering, perhaps with some shifting agriculture. Inevitably, this form of natural resource use (or lifestyle) requires large tracts of land per capita, and the inhabitants also usually lack formal “titles” to the land off which they live.
- 10 This model is simplified, but land is always heterogeneous, and some “centrally located” land may nevertheless be inferior for agriculture (because of slope, nutrient deficiency, rocky soil, poor drainage, or inaccessibility). In a more complex model, the protection of property rights to forestry may be either *cheaper or more expensive* than the corresponding protection of rights to agricultural land. Culturally and politically, it appears to be more difficult to defend property rights if land is not “used” (e.g., natural forest). Popular tradition is on the side of the squatters who work the land productively (Binswanger 1991); colonists in the Amazon often acquired both rights and subsidies through the act of deforestation.

Water

Demands on the world's available freshwater supplies continue to grow as the global population increases. During the 20th century, the global human population increased fourfold to more than six billion. In the same period, water withdrawn from natural freshwater ecosystems increased eightfold (Gleick 1998). In many parts of the world, scarce water supplies are a serious environmental concern, and water is often used at rates that exceed those dictated by the criterion of economic efficiency due to the lack of adequate policy, particularly with respect to pricing (e.g., Worldwatch Institute 1993). For example, in many of the formerly planned economies, water was provided essentially for free; as a result there was no culture of conservation.

Better management of water resources is essential to social and economic development, poverty reduction, and equity. Africa provides a good example since both growth and water are major challenges there. Investments in water infrastructure are required to meet basic needs in rural and urban areas and to enhance agricultural productivity. As development proceeds, water has to be better managed for energy and food production, transportation, and flood control.

From an economist's viewpoint, many challenges to designing and implementing water management systems arise, especially in developing countries. First, the provision of a water supply and sanitation network is a huge societal enterprise (Whittington et al. 2008). In industrialized and developing countries, the costs of reservoirs, canals, water transmission lines, urban distribution networks, pumping stations, water treatment facilities, sewerage collection, and wastewater treatment facilities often account for a substantial share of public sector investment.

Second, the provision of water and sanitation network infrastructure is very capital-intensive. Because so much capital is at stake, the property rights to the revenue from water and sanitation facilities must be well established and secured for private investors to feel confident in undertaking such large investments.

Third, household demand for small quantities of drinking water seems to be price-inelastic. For instance, in a meta-analysis of 124 estimates generated between 1963 and 1993 Espey et al. (1997) found a mean price elasticity in the

U.S. of -0.51 ; the short-run median was -0.38 and the long-run median was -0.64 , with 90% of all estimates being between 0 and -0.75 . When it comes to developing countries, things are more complex and varied. One reason is simply varying socioeconomic and ecologic conditions, but another is the dependence of so many households on multiple water sources, many of which are in the nonmarket sphere. A growing body of studies being published in this area providing a better understanding of household water use in these countries will be very important for management and expansion of their water systems (see Nauges and Whittington [2010] for a survey). Despite such heterogeneity, most estimates of own-price elasticity for private connections are within the range -0.3 to -0.6 , and thus they do not appear to differ markedly from those in industrialized countries.

Fourth, fixed costs for water systems are typically high whereas marginal costs are low (except when capacity constraints kick in), implying substantial economies of scale in several parts of the water supply chain. This provides arguments for public involvement since water supply may be characterized as a “natural monopoly,” and the low price elasticity introduces the risk that much money can be made if operators gain control of the capital assets and pursue an objective of maximizing monopoly profits rather than social welfare. The picture may be more complicated, however, since diseconomies of scale can exist in some parts of the water supply process too, thus making it difficult to judge the behavior of average costs. In some instances, arguments in favor of “natural monopoly” are absent.

Fifth, although water storage is relatively easy, long distance transportation is very expensive because water is so heavy. The geographical distribution of water contributes further to the management problem (see Table 23-1). Although ancient civilizations were created along many of the world’s major rivers, colonialism created national boundaries that often follow a completely different logic. Since it is very costly to import large supplies of water at short notice from other locations during droughts or periods of limited production capacity, demand management tools and pricing become necessary for coping with water shortages. Nevertheless, the high cost of transporting water combined with the inelastic demand imply that shortages can create desperate situations—especially in developing countries with little water storage and thus little protection against drought. In some parts of rural Tanzania, for instance, a 20-liter bucket of water can cost an unskilled laborer’s daily wage. The only options available may be to walk all day long for water or to work all day in the fields and buy water (Whittington et al. 2008).

Finally, water is distinctive because of its strong social, environmental, and cultural values. Literally, water is essential for life, and many believe that it should not even be considered an economic good at all but provided for free. (Already in the Bible there are complaints when people had to *pay* for water: Lamentations 5:4.) This argument may be morally compelling and politically attractive to some, but the sale of water below cost leads to a long-run lack

TABLE 23-1 Rate of Access to Safe Drinking Water by Country, 2004

<i>Region and Country</i>	<i>Urban</i>	<i>Rural</i>	<i>Total</i>
Africa			
Ethiopia	81	11	22
Ghana	88	64	75
Kenya	83	46	61
Somalia	32	27	29
South Africa	99	73	88
Tanzania	85	49	62
Uganda	87	56	60
Zimbabwe	98	72	81
America and Caribbean			
Bolivia	95	68	85
Costa Rica	100	92	97
Cuba	95	78	91
El Salvador	94	70	84
Haiti	52	56	54
Mexico	100	87	97
United States of America	100	100	100
Venezuela	85	70	83
Asia			
Afghanistan	63	31	39
China	93	67	77
India	95	83	86
Vietnam	99	80	85
Oceania			
Australia	100	100	100
Papua New Guinea	88	32	39

Source: WHO/UNICEF 2006. World Water Development Report.

of funds for system expansion and maintenance. If the municipal or private entities that provide the service receive very little revenue, the service ends up being inadequate unless subsidies from higher levels of government can be reliably secured. In the end, the poor individuals—not the rich—are most directly affected. The rich have means to purchase alternatives.

For drinking water, quality is important; contaminated water is a major source of disease and death in developing countries. Small quantities of high-quality water can be considered a merit good because some diseases often caused by poor-quality water (e.g., cholera) are contagious, and their costs affect all of society. Because poor people have few options but to drink

available water (often without further treatment) and because quality is not easily visible, water quality monitoring and public involvement are necessary. On the other hand immunization, for instance against cholera, is in a sense a partial substitute for cleaner water; immunization can similarly be discussed as a merit good since there are “herd protection effects” or positive externalities on neighbors of vaccination. In spite of this Cook et al. (2008) show that the case for subsidizing vaccines is not clear cut, and provision at market cost may be welfare-improving compared to exaggerated subsidies.

Many places around the world saw a wave of municipal water supply privatization in the 1990s and early 2000s. Although the principles of market economics and charging consumers according to cost are important, privatization of water supply is not necessarily always beneficial. For such arrangements to be socially acceptable, they need to be governed by careful legislation and contracting that details the characteristics of service expected.

Water Management and Tariff Design

The economic costs of providing water and sanitation services to households are the sum of several components (Whittington 2011). These include: opportunity costs of diverting raw water from alternative uses to the household; storage and transmission of untreated water to urban areas; treatment of raw water to drinking standards; distribution of treated water within the urban area to the household; sewerage collection; and treatment and costs or damages imposed on others by the discharge of wastewater (negative externalities). Some cost components are subject to significant economies of scale—for example, storage and transmission, treatment of raw water to drinking standards, and sewage treatment. Quite the opposite, others are subject to diseconomies of scale—that is, the unit cost increases with the quantity of water consumed. For example, the search for additional freshwater supplies becomes harder and harder as large cities become larger.

Setting water tariffs requires striking a balance between several objectives. If economic efficiency is pursued, the volumetric charge should be set equal to the short-run marginal social cost of supplying one additional cubic meter of water into a city. On the other hand, investments in water systems are typically large and indivisible, and there is an important distinction between short- and long-run marginal costs. In the short run, supply (and demand) may be inelastic, with low prices because all the fixed costs of the system are “sunk costs” and the marginal costs small (Point A in Figure 23-1). When demand grows as income and population grow, exceeding the supply capacity of the current system, expensive new investments might be necessary to provide the necessary supply. The intersection of the long demand D_{lr} with the long-run marginal cost of water supply LRMC is denoted by B in Figure 23-1. The long-run marginal cost lies far above the corresponding short-run curve:

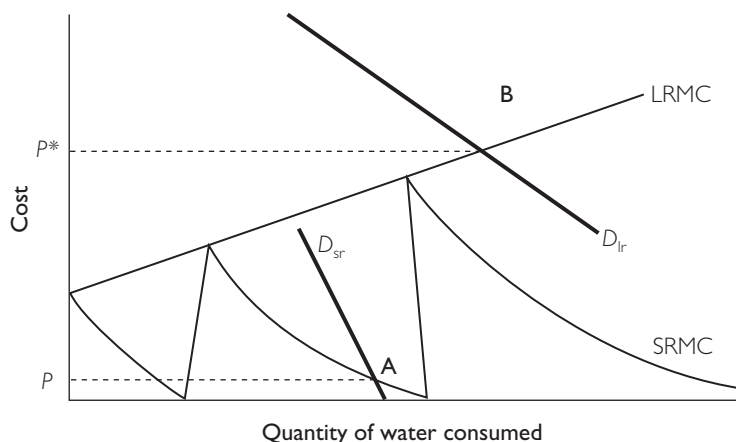


FIGURE 23-1 Short- and Long-Run Costs

Notes: P and P^* = short-run and long-run equilibrium price, respectively; D_{sr} and D_{lr} = short-run and long-run demand, respectively; A and B = short-run and long-run equilibrium, respectively; SRMC and LRMC = short-run and long-run marginal cost, respectively.

somehow we need to finance the costs represented by the shadow price at B in Figure 23-1. Therefore, cost recovery requires that on aggregate, tariffs faced by consumers should produce revenue equal to the financial costs of supply. If this is not the case, incentive to invest in new capacity will be lacking, forcing such capacity to be financed by some other means. On the other hand, new capacity can to a large extent be financed by fixed fees so as not to distort the relationship between the short-run marginal cost and the price paid at the margin.

The main concern with a market-based approach to urban water management arises from the fact that the allocation of a scarce good in the market depends on the willingness to pay (WTP). Since a portion of water in residential consumption is used for basic needs, policymakers should be concerned about the risk that a particular tariff structure will make it impossible for very poor households to get a sufficient quantity of water to maintain public health. However, it is not clear that command-and-control policies are more effective (Olmstead and Stavins 2009). Water tariffs can, however, be designed that reduce the water bills of poor households without subsidizing usage (Boland and Whittington 2000).

The concepts of unitary, progressive, and regressive (efficiency) tariffs discussed in Chapter 21 for electricity also apply to water. The tariff structure of choice in developing countries (Boland and Whittington 2000) has become the increasing block tariff (IBTs)—a volumetric tariff where a water user in a particular category (e.g. residential) is charged a unit price for the first units abstracted, up to a specific amount that defines the end of the initial first block; above this amount the user faces a higher price for additional units until the second block, and so on until the top block. The IBT adopted by

the municipality of La Paz, Bolivia, in conjunction with the local water utility (SAMAPA) and the Bolivian national tariff board in 1997 is a good example. First, residential users face large price differentials between blocks. In this case, the price in the most expensive block was over five times the price in the least expensive block. Second, domestic water users attract most of the attention of tariff designers; the residential use category contains more blocks than the commercial or industrial categories. Third, prices charged to industrial and commercial users are much higher than those applying to typical levels of residential water use.

Proponents of IBT structures make several arguments. First, it is claimed that IBTs promote equity because they force wealthy households to cross-subsidize poor households. The low price in the initial block means that the poor can obtain a sufficient quantity of water for their essential needs at a low monthly cost. High-income households are supposed to pay a higher average price for water because a greater percentage of their water use occurs in the higher price blocks. Second, it is also claimed that the high rates charged to industrial and commercial customers relative to most residential customers promote “equity” by allowing the water utility to cross-subsidize poor residential customers with revenues from industrial firms. Third, the prices associated with the highest blocks can discourage or stop “extravagant” or “wasteful” water use, promoting conservation and sustainable water use. Another claim is that IBT can mimic a rising marginal cost curve. If the price of water in the most expensive block is set equal to marginal cost, then marginal cost pricing can be accomplished. The argument that IBTs help poor households is based on three assumptions:

- 1 Poor households are connected to the piped systems;
- 2 Poor households always use less water than rich households; and
- 3 Rich households pay more than the long-run average costs of supply.

All three of these assumptions are often violated in practice, with the result that the vast majority of the subsidies typically go to middle and upper income households.

The political reality of most tariff setting procedures means stakeholders or consultants rarely participate in the process or see the financial or distributional consequences of expanding the size of the initial block. The same political realities make it difficult to restrict the size of the middle blocks as well.

Experience has shown that with efficient management and an appropriate tariff structure, urban systems may even be run profitably in middle and high income countries. One example is the water enterprise in Santiago, Chile (Dixon and Howe 1993), where an IBT system has been in place since 1989, complemented with an explicit consumption subsidy. This subsidy is targeted on the poorest 20% of the population and is higher in tariff zones where water charges are higher and average income of the poorest population is lower.

On the other hand, China has applied a system of unitary water tariffs across the country. In the United States, some municipalities have used a *decreasing* block structure. This approach is a better approximation of cost structure but has been criticized for benefiting the large consumers and leading to waste, and lately some utilities have adopted increasing block rates instead.

Water Management: Nonprice Approaches

Although theoretical and empirical economic analyses suggest that market-based policies are more cost effective than command-and-control (CAC) approaches, policymakers and water suppliers have historically relied mainly on CAC conservation programs to manage water resources and to induce demand reductions during shortages, mainly due to distributional concerns (Olmstead and Stavins 2009). Three main types of programs have been applied: (1) required or voluntary adoption of water-conserving technologies; (2) mandatory water use restrictions; and (3) mixed nonprice conservation programs.

Many urban water utilities have experimented with required or voluntary adoption of low-flow technologies. For instance, the U.S. Energy Policy Act required the installation of low-flow toilets and showerheads in all new residential construction; some cities have also mandated or encouraged retrofitting. However, studies of the water savings induced by such policies have varied widely—from zero to significant water savings—because the policies induced some unexpected behavioral changes that partially offset the savings benefits. In the United States, for example, households with low-flow showerheads seem to take longer showers than they would without these fixtures. On the other hand, households fully retrofitted with low-flow toilets used about 20% less water than households with no low-flow toilets (Mayer et al. 1998). Evidence shows that these can be expensive programs in terms of the cost per liter saved (Olmstead and Stavins 2009).

Much like the traditional CAC approach to pollution regulation, mandatory water use restrictions on the total quantity of water as well as restrictions on particular water uses have been implemented in some countries. For example, the “big dry” that has affected much of southeast Australia since 2001 has reduced the water in storage in many locations in that country. State governments and water utilities have used mandatory water restrictions to reduce demand by banning various outdoor water uses. As of March 2008, at least 75% of Australians live with mandatory water restrictions.

Although water utilities and state governments are aware of the economic inefficiencies of rationing, various justifications for rationing water instead of charging a higher volumetric price may be used:

- 1 If water is considered a basic need then allocating it on the basis of price could be considered inequitable because it places a large cost burden on

- poorer households; a simple solution, however, is to give a fixed rebate on the water bill (see Boland and Whittington 2000).
- 2 Second, in some communities, household water consumption is not metered. Thus, raising the water price in the form of a fixed charge provides no financial incentive to consumers to reduce their demand.
 - 3 Third, even when households are metered and are charged a volumetric price for their water, the billing period may be, for instance, quarterly; if an immediate and temporary reduction in demand is required, it may be more effective to implement a rationing scheme than to raise the price.

However, the welfare losses of rationing might be considerable. For example, Grafton and Ward (2008) estimated the aggregate daily water demand for Sydney and used it to calculate the difference in households' surplus between using the metered price of household water to regulate total consumption versus mandatory water restrictions for the period 2004–2005. The loss in consumer surplus from using mandatory water restrictions was calculated to be \$235 million. On a per capita basis this equates to approximately \$55 per person, or about \$150 per household per year—a little less than half the average Sydney household water bill in 2005.

Finally, water policies must combine many elements, from rationing to publicity and education campaigns imploring consumers to use water wisely, making it difficult to determine the effects of individual policies. Renwick and Green (2000) assessed the relative performance of alternative policy instruments in reducing aggregate demand in California. Their results indicate that CAC policies reduce aggregate water demand. However, the characteristics of the instrument influence the outcome of these policies. More stringent mandatory policies such as use restrictions and water allocations reduce aggregate demand more than voluntary measures such as public information campaigns and retrofit subsidies.

Behavioral factors also play a role. For instance, Millock and Nauges (2010) assess the relative impact of attitudinal and behavioral variables (among others) on household adoption of water-efficient equipment. They find that a strong commitment to environmental values increases the probability of adopting indoor water-efficient equipment by 9% on average—a similar order of magnitude as price variables. Nevertheless, in many cases, solutions to environmental problems may require combinations of policy instruments (see Chapter 15). In the following sections we discuss some concrete country examples—not necessarily because they are unique to the countries, but rather the opposite: many of them are fairly typical for many countries.

Tariff Structures in Some Middle Eastern Countries

Decisionmakers in many countries appear to be convinced that because the aggregate supply curve for water slopes upward in the short run, so should the

individual tariff, but this is not necessarily so—at least not in terms of allocative efficiency. If a country faces (increasing) scarcity of water, then the tariff levels should be appropriately high (and maybe rising over time). Neither surface water sources nor groundwater aquifers can discriminate between a small consumer consuming his first gallon and a large consumer consuming an additional gallon. There is thus no technical efficiency argument for IBTs.

Given the importance of water for health and welfare, it may be appropriate in some contexts to have a subsidized “lifeline block.” The size of the volumetric charge in this lifeline block should logically be tied to the volume of water consumed for personal use (i.e., washing, drinking, and cooking)—on the order of 10–20 liters/day per capita. Many times the size of the lifeline block is made too large. Saudi Arabia is an extreme example, where tariffs were set to almost zero ($< \text{US\$}0.05/\text{m}^3$) for a monthly consumption of up to 100 m^3 , which translates to use of more than 3,000 liters/day. Presumably, most families are on the first step of the tariff, which in practice removes the progressivity (see Figure 23-2). Even so, in many cases, the poor consumers who are supposed to be the main beneficiaries of the lifeline and increasing block tariffs do not have permanent, individual household connections and end up consuming poor-quality water collected locally or water purchased from water vendors at extremely high rates.

Geographical location and season are often the main determinants of water scarcity and thus the marginal cost of providing water. Reflecting these variations in tariffs can increase overall allocative efficiency. In Kuwait and the United Arab Emirates, (unitary) tariffs are high because actual costs have been allowed to influence tariffs to a large extent. This gives some perspective on the subsidies implicit in the tariffs in Saudi Arabia. Even so, water experts at the World Bank indicate that Saudi Arabia appears to be modifying the “lifeline”

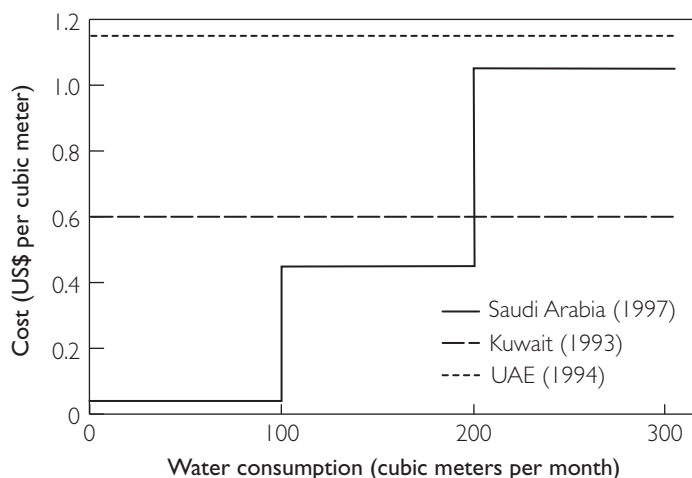


FIGURE 23-2 Water Tariffs in Saudi Arabia, United Arab Emirates (UAE), and Kuwait

rate by reducing the maximum monthly water consumption at the extremely low tariff (World Bank 1997b).

Water Tariffs in Chile

In recent decades, Chile has pursued neoliberal, market-oriented policies in the water sector. Chilean authorities have set out to privatize previously public water management services. The argument is that the role of the state should be to regulate services, not provide them. Currently, only a small percentage of urban water and sanitation services are provided by private enterprises. Rather, municipal water services are commonly provided by public corporations that operate as independent legal bodies.

Departing from the more traditional rate-of-return and price-cap regulations, prices of public water utilities in Chile are set using a particular form of yardstick regulation in which the benchmarking is based on a hypothetical efficient firm (Montero 2005). The tariffs are based on calculations for “model enterprises” that allow for cost recovery and profitability (7% return on capital) for efficient firms and must avoid cross-subsidies between user categories. The recommended tariff structures include fixed charges for fixed costs (such as metering), capacity charges (depending on seasonal costs), consumption charges, and charges for waste treatment. All of these charges vary by region (and season, where applicable), depending on the actual levels and costs of water supply, infrastructure, treatment, transport, sewage, and so forth.

When the new tariff structures were applied beginning in 1990, the rate increase was particularly dramatic for the low-volume consumers, who previously had the most heavily subsidized rates (e.g., more than 450% for users of <20 m³/month in the Atacama Region). To soften the blow, the reform was implemented slowly over four years. In addition, a system of subsidies still exists for consumers who are unable to afford their water bills. To achieve transparency in public affairs, this subsidy is not automatic in the form of a reduced “lifeline” tariff but is a direct, means-tested subsidy. To receive this subsidy, consumers must submit a written application to the local municipality explaining their inability to pay. Successful applicants must meet several conditions. For example, one condition for eligibility is that a water bill for 20 m³ must cost more than 5% of the household’s income.

In 1995 more than 15% of customers received subsidies. This subsidy arrangement effectively targets poor individuals while preserving transparency in the public use of money. However, the system does incur administrative costs for the public agencies and time costs for poor consumers, and it may be somewhat humiliating for those who are forced to apply for the subsidy.

Water Management, Laws, and Pricing in Southern Africa

Southern Africa experiences considerable water stress. Large areas are arid or semi-arid. Evaporation is high, and rainfall is moderate and erratically distributed over time and space. These characteristics make it important to organize the storage, transport, and general management of water. Requirements for these functions are radically different in Botswana and South Africa. Whereas Botswana is mainly a rural, pastoralist society, South Africa is a highly diverse and complex society with many competing users.

Despite the strong position of landowners in South Africa¹, riparian rights do not enjoy a strong legal standing (see Chapter 3). According to the 1998 National Water Act, all water rights are owned by the state; however, ownership and user rights are distinct. The legislation also recognizes some historic private user rights. In some respects, the catchment management associations and water users associations strengthen such private user rights through extended stakeholder participation in the allocation of user rights. However, the law states explicitly that water rights do not stem from the ownership of adjacent land (contrary to the riparian principle).

Decisionmakers have made a distinction between water resource management, which is regulated by the Water Act, and the provision of water, which is regulated by the Water Services Act. The Water Act creates new rights based on the principles of equity, sustainability, and efficiency. Under the equity principle, the act recognizes basic human water needs as a right—in practice, each person is entitled to 25 liters/day, irrespective of income. Poor consumers pay nothing up to this level. Under the sustainability principle, the act recognizes the need for an ecological reserve, sometimes referred to as the “instream flow requirement,” that is not available for any economic use. Because the ownership of water is ultimately public and because ecological needs are a first priority, restrictions may be placed on land use and other activities that have effects on water availability or quality. One example is the restriction on planting trees, due to their demand for water (see Box 23-1). Planting trees in arid areas increases evapotranspiration and thus typically reduces surface water flows. (“Green” NGOs and others often forget this and simply argue that afforestation always increases water availability.)

Because of the rights created to provide for basic human water needs, the state is obliged to organize water supply, particularly in impoverished regions. The geophysical conditions of the country make this task expensive; they put considerable strain on already insufficient public funds and thus necessitate the involvement of the private sector through privatization or other arrangements. Although the Water Act promotes the role of the state as custodian of water related to basic human needs and ecological needs, the Water Services Act of 1997 strives for efficiency and makes provisions for privatization and competition at various levels along the water supply chain. Public–private partnerships are a new element of water management in South Africa (Conradie et al. 2001).

BOX 23-1 Prize-Winning Water Policies

Professor Kader Asmal, former minister of water affairs and forestry in South Africa and chairman of the World Commission on Dams, was awarded the 2000 Stockholm Water Prize for his extraordinary efforts in the field of water management in South Africa.

When Asmal was appointed to the ministry in 1994, more than 16 million South Africans did not have reasonable access to safe drinking water. In 1995, he created Working for Water, a water conservation program intent on removing invasive alien plant species that rob South Africa of up to 7% of its mean annual runoff, overtake its most productive lands, and threaten its biological diversity. He encouraged major water reforms in the 1998 National Water Act and the Community Water Supply and Sanitation Program, which focuses on providing access to the basic levels of service required to ensure health for all South Africans. More than 7 million people have already benefited, directly or indirectly, from these innovative programs.

One of South Africa's water management policies is aimed at controlling the planting of trees. Any commercial planting requires a license from the Department of Water Affairs and Forestry for what is called a "stream flow reduction activity." The licensing process also involves the Department of Agriculture and the Department of Environmental Affairs in a holistic approach to water management.

Sources: All Africa 2000; SIWI 2000

Pricing Water When Metering Is Not Possible

Pricing water for agricultural use is often difficult because of riparian rights or practical reasons.² It is nonetheless desirable when water is scarce. Irrigation water is commonly free or calculated based on the acreage irrigated. However, such a pricing scheme would not normally create the correct incentives to economize water. In some cases, technological progress in metering can reduce monitoring costs significantly, changing the conditions on which the choice of instruments hinges. (For example, time-of-day meters for electricity and on-board or curbside monitoring of air pollutants from cars were unimaginable just decades ago.) The sociocultural context is also crucial. In a closely knit society, monitoring by neighbors may be a viable alternative to monitoring by more distant central authorities or companies. In this case, common property resource (CPR) schemes may have comparative advantages.

A couple of options may still be considered. The first option is partial or voluntary metering. Requiring a category of users to install meters may be possible if they are distinguishable as particularly intensive (or wasteful) water consumers or if other special characteristics such as location apply. Second, offering some water users the option of metering may be successful if the alternative is a fairly drastic fixed charge. The third option is policy instruments, which may target the acquisition of certain types of capital (water-saving or water-wasting) equipment. Finally, authorities may regulate the choice of crop

and cultivation methods, which are more readily observable than water use. They would then design the charge so as to use the “revelation principle” (see Chapter 10), allowing water authorities some control over irrigation use even without metering. Designing such tariffs is mathematically rather complicated. However, the approach is of great general interest because it could be applied to pollution charges or other natural resources fees. We illustrate the method for one concrete example, irrigation water.

Assume that the water authority knows all farmers have a productivity “type” β (depending on slope, crops, soil quality, and maybe skills) and that β has a rectangular distribution over $(0,1)$ such that the farmers’ harvest (q) will be a function of water use (w), as in

$$q = (1 + \beta)w^{0.5} \quad (23-1)$$

Assume that the farmer has private costs of irrigation and pumping (c) per unit of water and the price received for each unit of harvest is P . The farmer will then maximize private profits (π) as

$$\pi = Pq - cw \quad (23-2)$$

The water, however, has an external scarcity-related value, e . Social welfare considerations would thus be attained by maximizing the social profit function (π^*) as

$$\pi^* = Pq - (c + e)w \quad (23-3)$$

Inserting Equation 23-1 into Equation 23-3 and differentiating gives

$$\pi^{*'} = 0.5P(1 + \beta)w^{-0.5} - (c + e) = 0 \quad (23-4)$$

Maximizing gives

$$W^* = P^2(1 + \beta)^2/4(c + e)^2 \quad (23-5)$$

$$q^* = P(1 + \beta)^2/2(c + e)^2 \quad (23-6)$$

where q^* and w^* are the socially optimal values for production and water use, respectively, and are distinct from an open access outcome (q_m and w_m , respectively). Open access values q_m and w_m are calculated by using Equations 23-5 and 24-3 with $e = 0$. With full information, the traditional Pigovian instrument would have been a tax of e per unit of water. Tradable water rights with a price of e would have worked equally well.

If monitoring is reasonably inexpensive, then the best policy is to require metering. However, if distributional concerns and political factors block any

reform that attempts to make water meters compulsory, then it may be in the interest of the entire farming community to block or fight the installation of meters. A decentralized policy with somewhat better chances of successful implementation might be the imposition of a fixed fee on each user for the total scarcity cost combined with an exemption for users who adopt water meters (i.e., metered users would pay for actual water consumption). Large-volume consumers typically prefer a fixed charge, whereas low-volume consumers typically prefer monitoring. In general, when compulsory metering is dominated by partial adoption of meters by a subset of agents, a policy of decentralized monitoring can improve social welfare (e.g., Millock et al. 2002). The size of the fixed fee is crucial. A low fee will lead only a small fraction of users to choose to monitor (thus, only that fraction will select optimal water use and production levels). However, a high fee will invite political resistance and perhaps questions concerning the feasibility of the policy. Imposing a high fee to encourage the use of water meters relies on two strong assumptions: the water authority's threat of a fixed charge is credible, and there are no transaction costs from taxation.

If these issues cannot be resolved or if monitoring is physically difficult, regulating water use in a welfare-maximizing manner would seem to be an impossible task. However, possibilities exist for water authorities who have information that is correlated to water use. Assume, for instance, that the water authority can monitor output perfectly and is the sole buyer of the crop (e.g., through an agricultural marketing board). The water authority could then offer a menu of two-part contracts with different harvest levels and water fees designed so that all farmers reveal their type (β). The contracts $[T(\beta), Pq(\beta)]$ would consist of a fixed water charge (or entry fee into the scheme), $T(\beta)$, and an offer to buy an output (harvest) that depends on each farmer's type, $q(\beta)$, at a price P . (In excess of this level, there would be no payment).

The revelation mechanism hinges on the existence of two components in the contract. Farmers might be tempted to declare a low β to get a low water charge but then would have the right to produce and sell only a small harvest $q(\beta)$. The calculation of the fee scheme $T(\beta)$ that satisfies incentive compatibility conditions 23-8 and 23-9 is given in Box 23-2. If the private cost of pumping water $c = 1$, harvest price $P = 5$, and the external scarcity rent of water $e = 0.5$, then $T(\beta) = (25/9)(2 + 2\beta + \beta^2)$ (see Equation 23-10). Using this scheme, the payoffs can be calculated for farmers of different true types (β_i) if they claim to be other alleged types (β_a).

If a farmer reports $\beta_a > \beta_i$, then the marginal tax rate is higher than marginal gross profits (before tax). If a farmer reports $\beta_a < \beta_i$, then the marginal tax rate is lower, but then his or her allowable harvest is lower, which has a stronger effect on profits (see Table 23-3). For a farmer of type 1, the socially desirable production level (according to Condition 23-6) would be 20/3, or 6.7 units. The water authority will offer a contract intended for this type. A farmer of type 1 who agrees to this contract will be allowed to sell 6.7 units, giving a

BOX 23-2 Design of Self-Revealing Tariffs

If the range of technologies (Equation 23-1) is known, then contracts can be designed such that each farmer pays a fee (T_β) for the right to sell harvest (q_β). With increasing farm quality (β), both q_β and T_β will rise. The water authority sets q_β equal to the level that would be optimal if β were truthfully reported [i.e., $q^*(\beta)$]. The issue then becomes how to design the fees. The water authority knows the technologies and costs and can thus simulate what each farmer of type β would do given his or her true type. The water authority must choose the fee so as to make each farmer pick the contract corresponding to his or her type. Note that profits are gross profits minus the fee, $T(\beta)$, and can thus be written as

$$T(\beta) = Pq(\beta) - cw[q(\beta)] - \pi(\beta) \quad (23-7)$$

where P = harvest price, c = private cost of pumping water, w = water use, and π = profit.

Consider a farmer of true type (β_t) contemplating the false reporting of alleged type (β_a). Because the allowed quantity of production and tax rate depend on the type reported, the farmer will report β_a to maximize profits. The first-order condition for the farmer is

$$\frac{\partial \pi}{\partial \beta_a} = P \frac{\partial q(\beta_a)}{\partial \beta_a} - c \left(\frac{\partial w}{\partial q} \right) \left(\frac{\partial q}{\partial \beta_a} \right) - \frac{\partial T(\beta_a)}{\partial \beta_a} = 0 \quad (23-8)$$

The tax schedule $T(\beta_a)$ must be made to (at least) match the increased revenue due to false reporting. For the planner, the marginal social profit is

$$\frac{\partial \pi}{\partial \beta_t} = -c \left(\frac{\partial w}{\partial \beta_t} \right) \quad (23-9)$$

If $\beta_a = \beta_t = \beta$, then $\partial \pi / \partial \beta = P \partial q(\beta) / \partial \beta - c(\partial w / \partial q)(\partial q / \partial \beta) - c(\partial w / \partial \beta) - \partial T(\beta) / \partial \beta = -c(\partial w / \partial \beta)$ because $P \partial q(\beta) / \partial \beta - c(\partial w / \partial q)(\partial q / \partial \beta) - \partial T(\beta) / \partial \beta = 0$ from Condition 23-8. This is the envelope condition for truth-telling and is an empirical counterpart to Conditions 10-3 and 10-4 in Chapter 10.

The design of the optimal tariff can be illustrated using a numerical example in which $c = 1$, $P = 5$, and the external scarcity rent of water (e) = 0.5. It gives optimal values for harvest and water cost of $q^* = (5/3)(1 + \beta)^2$ and $w^* = (25/9)(1 + \beta)^2$, respectively. For the farmer who cheats, however, the values for harvest and water cost are $q(\beta_a) = (5/3)(1 + \beta_a)^2$ and $w(\beta_a, \beta_t) = [q(\beta_a)]^2 / (1 + \beta_t)^2 = (25/9)(1 + \beta_a)^4 / (1 + \beta_t)^2$. Equation 23-9 gives

$$\frac{\partial \pi}{\partial \beta_t} = 2c \frac{[q(\beta_a)]^2}{(1 + \beta_t)^3} = \left(\frac{50}{9} \right) \frac{(1 + \beta_a)^4}{(1 + \beta_t)^3} = \left(\frac{50}{9} \right) (1 + \beta)$$

when $\beta_a = \beta_t = \beta$. By assuming a uniform distribution of $\beta \in [0, 1]$, integrate over β to calculate the profit (π):

$$\pi = \int_0^1 \left(\frac{50}{9} \right) (1 + \beta) d\beta = \left(\frac{25}{9} \right) (2\beta + \beta^2)$$

Hence, from Equation 23-7, the optimal tax rate is

$$T(\beta) = \left(\frac{25}{3}\right)(1+\beta)^2 - \left(\frac{25}{9}\right)(1+\beta)^2 - \left(\frac{25}{9}\right)(2+2\beta+\beta^2) \quad (23-10)$$

To verify that this tax forces the farmer to tell the truth, calculate the marginal profitability of lying about β from Equation 23-8:

$$\begin{aligned} \frac{\partial \pi}{\partial \beta_a} &= \left[5 - \frac{2q(\beta_a)}{(1+\beta_t)^2} \right] \left(\frac{10}{3} \right) (1+\beta_a) - \left(\frac{25}{9} \right) (2+2\beta_a) \\ &= \left[3 - \frac{2(1+\beta_a)^2}{(1+\beta_t)^2} \right] \left(\frac{50}{9} \right) (1+\beta_a) - \left(\frac{50}{9} \right) (1+\beta_a) \\ &= \left(\frac{100}{9} \right) (1+\beta_a) \left[1 - \frac{(1+\beta_a)^2}{(1+\beta_t)^2} \right] \end{aligned}$$

Setting $\partial \pi / \partial \beta = 0$ as a maximum condition gives $(1+\beta_a)^2 / (1+\beta_t)^2 = 1$, and thus $\beta_a = \beta_t$.

This result shows that profit is (now) maximized by truthful reporting of β_t .

TABLE 23-2 Truthful Revelation of Types Applied to Water Contracts

True type	Alleged type		
	$\beta_a = 0$	$\beta_a = 0.5$	$\beta_a = 1$
Contracted quantity	1.7	3.7	6.7
Revenue	8.3	18.7	33.3
Tax $[25/9(2 + 2\beta + \beta^2)]$	5.5	9.0	13.9
<i>Gross profit (net profit)</i>			
$\beta_t = 0$	5.5 (0)	4.7 (4.3)	11.1 (25)
$\beta_t = 0.5$	7.1 (1.6)	12.2 (3.2)	11.1 (2.8)
$\beta_t = 1.0$	7.6 (2.1)	15.2 (6.2)	22.1 (8.2)

Notes: β_t = true type; β_a = alleged type. Numbers refer to the example in the text. Net profit = gross profit minus tax.

gross revenue of 33.3 and a gross profit of 22.1; the farmer will pay the fee of 13.9 and have a net profit of 8.2. If instead the farmer claims $\beta_a = 0.5$, the water fee would be 9.0 instead of 13.9, but the farmer would have the right to sell only 3.7; therefore, net profit would fall from 8.2 to 6.2. Other types can be verified similarly.

The presence of transaction costs in raising tax revenue complicates the issue further, introducing informational rents and making the attainment of a “first- best” solution impossible (Smith and Tsur 1997). If transaction costs are

too large, then society may be better off without regulation. Regulation, and particularly regulation with transaction costs, generally appears to decrease the desirable level of production from farmers with low-quality type. This effect may raise distributional concerns.

This model is sensitive to changes in specification. Suppose, for instance, that there is some uncertainty about the exact shape of the function (see Equation 23-1); it might instead be $q = (1 + \beta)w^{0.6}$ or some other small deviation. Changing the elasticity estimate might be in the interest of low-output farmers (on poorer soils) who want to prove that they are not inefficient. Other changes in the formula would benefit other categories. If considerable amounts of resources depend on such numbers, then lobbying or rent-seeking is expected.

Notice also that the contracts designed do not provide incentives to economize on water after the contract is signed. The whole effect on water is indirect, working through the optimal output levels. Therefore, interactions with other policies might be expected. Many policies already target output levels: export promotion, agricultural support, poverty relief, crop insurance, income tax, and property tax, to name a few. When the policy proposed is implemented in addition to other such policies, the net effect may be blurred and even unpredictable. In the example above, this effect is partly indicated by the rising transaction costs. An increasing divergence between private costs (c) and social costs ($c + e$) also would tend to exacerbate the problem. In several cases, c may in fact be close to 0, making the role of e relatively more important.

The model presented above is highly stylized. That is the function of models—otherwise, cases cannot be analyzed. In this case, we assume that the water authorities have no information about individual water use but full knowledge about all technologies as well as full visibility and control of crop output. In real life, crops generally depend on stochastic factors, adding moral hazard. It is not difficult to imagine that different crops have different monitoring costs. Subsistence farming would require a good deal more monitoring than would export crops such as coffee or cotton, which are sold to centralized agents, marketing boards, or factories. If some farmers were growing cotton or subsistence crops and the poorer marginal farmers had difficulty growing the export crop without large quantities of scarce water, then a simplified two-price mechanism might build on discrimination between crops and farm types. By requiring signed contracts *ex ante* with farmers for cotton and specifying an “entry fee” (as a proxy for the unobservable use of water) for such contracts, farmers with superior conditions for the relevant crop would self-select without interfering with the production of the subsistence crop. Water monitoring is often expensive, and in these cases, separating equilibrium may be interesting where the authorities charge for water based on acreage and crop. By providing farmers incentives for growing less water-intensive crops, the policies will have at least some effect on total water use.

One more category of situation should be mentioned. Other kinds of capital equipment may be needed for production that provide new possibilities

to policymakers. The production function might be more complicated than Equation 23-1, and complementarities between capital and water may give valuable information. At least in the 1990s, French water agencies designed contracts to make firms extend their wastewater treatment more than motivated by the current tax, which was fairly low (A. Thomas 1995; see also Chapter 10). To set the correct abatement requirements for each firm, water authorities needed information about each firm's individual abatement costs, which was effectively acquired by offering firms a subsidy for this pollution control equipment.

CPR Management of Water

CPR management systems for collective property rights can be seen as policy instruments through which users of natural resources at the local level manage common property resources under difficult circumstances related to the stochasticity of natural conditions (see Chapters 3, 8, and 10). Irrigation schemes usually are used to illustrate CPRs.

In the Spanish towns of Alicante, Murcia, Orihuela, and Valencia, social organizations have evolved for regulating irrigation for the small-scale farms called *huertas* (Ostrom 1990). Rainfall is low and erratic in this warm, dry region, and the water level in rivers is variable. Thus, the success of agriculture is wholly dependent on irrigation. Over many hundreds of years, farmer organizations have evolved in response to the stochastic and somewhat harsh environment. Written records go back about 600 years, but the many Islamic influences in the procedures indicate that irrigation associations were in place several hundred years earlier, under Muslim rule.

One of the characteristic features of the *huerta* irrigation organizations is the *Tribunal de los Aguas* (Water Court). In Valencia, for many centuries it met every Thursday outside the Apostles' Door of the cathedral. The Tribunal has never been an official part of the Spanish judicial court system, but its jurisdiction over the management of the irrigation canals is nonetheless respected. The governing officials of the Tribunal are democratically elected by the farmers. The officials hire guards and have the power to allocate water. The main allocation mechanism for irrigation water is a physical system of rotation. Each farmer is allocated water in turn and can take as much as is wanted (except in periods of water shortage, during which the chief official or syndic can order additional rationing). A farmer who misses a turn by failing to open the gate when the water arrives must wait an entire rotation for another turn.

Under these circumstances, the farmers automatically monitor each other as they wait for their turns; however, the temptation to take water out of turn may be extremely high. The main tasks of the court are to solve conflicts concerning the incorrect taking of water and sometimes allegations of incorrect management by the syndic or his or her employees. Carefully graduated

finest appear to play an important role in maintaining the integrity of the institution and avoiding the temptation to free ride. This longstanding practice was recorded in the fine books as far back as 1443, when more than 400 fines were collected in one year.

Details of *huerta* irrigation management in Alicante, Murcia, and Orihuela differ in numerous details from that of Valencia, showing the importance of adaptation to local conditions, both ecological and social. Only through this kind of adaptation are the motivation, credibility, authority, and flexibility of the systems maintained (see Chapter 8). Other irrigation schemes, such as the Zanjera irrigation communities of the Philippines (Ostrom 1990), have some elements in common with the *huerta* systems, perhaps due to the transfer of some experience in combination with Spanish colonization. However, the ecological and technical conditions are distinct.

Water scarcity is not such a prevalent problem in the Zanjera irrigation communities. Instead, the rivers are prone to excessive water flow and flooding, which destroy dams and other simple irrigation constructions built by the shareholder farmers. Farmers therefore spend 40–50 days of labor per year on heavy and sometimes dangerous repair and maintenance work. The temptation to shirk collective repair work and thus free ride on the efforts of others is perhaps the greatest challenge to be overcome in such a system—more important than the allocation of water in times of drought. The institutions are accordingly focused on the issue of collective labor supply for maintenance.

This chapter is not easy to summarize; water can be as elusive as it is important. It poses challenges when it comes to drawing up ecosystem boundaries and legal principles. It is an area where the ideology of free access for all has a firm foothold, but we know in reality that it is unrealistic to base serious allocation decisions on the free provision of expensive goods or services. The extent of provision, the setting of tariffs, and the organizational structure of the water industry are all contentious and complex areas. No standard answers or “one-size-fits-all” remedies exist, but we must strive to use welfare theory to combine efficiency with social and ecological constraints that vary between different uses and different cultures and ecosystems. We have illustrated this with discussions that range from different kinds of simple or complex tariff design to common property resource management.

Supplemental Reading

Houston and Sun 1999

Millock and Nauges 2010

Nauges and Whittington 2010

Olmstead and Stavins 2009

Sanford and Stroud 2000

Whittington 2011

Wood, Handley, and Kidd 1999

Notes

- 1 This text is based mainly on Conradie et al. 2001 and Department of Water Affairs and Forestry 1996. Thanks to Martine Visser for valuable comments.
- 2 This section is based on Smith and Tsur 1997 (see also Millock and Salanié 1997; A. Thomas 1995). Thanks to Katrin Millock for good discussions on this topic. This section contains more mathematical detail than other sections to show how a self-revealing tax can be constructed; the most technical part is presented in Box 23-2.

Waste

For many policymakers, waste management is the epitome of environmental management. The dumping of solid wastes and the flow of untreated effluents can disrupt sensitive ecosystems, leading to the deterioration of water quality and thus poor human health. The handling of hazardous waste is especially worrying, given the developed countries' unfortunate trend toward dumping their wastes in developing countries. Related issues include the health problems of people who work in (and, in poor countries, even live on) waste sites.

Much effort is devoted to waste management, but sometimes the policies being pursued appear to be more symbolic than effective. In many countries, a large amount of political effort is devoted to promoting recycling. Germany is a good example, where extensive legislation requires the recycling of various categories of waste. It seems, however, that not enough effort has been devoted to the issues of what should be recycled, why, and which policy instruments should be used. The price mechanisms on the markets for used goods must be understood in order to discuss recycling. Some products have a high price, so the market "recycles" them automatically. Consider gold: have you ever heard of piles of gold (or other precious metal) waste? If a good becomes scarce, its price rises, creating incentives for reuse and recycling (see Chapter 22). Typically, this incentive does not apply to goods that can be produced industrially, including many minerals that are industrially excavated and processed. For many goods, the normal process of technological progress results in a decreasing price, at least measured relative to average income. A common description of a prospering economy is that income grows when measured at constant prices. Another way of describing it would be to keep income constant and say that the (income-deflated) prices are falling.

Some observers misinterpret this development as being primarily a change in attitude. They lament the fact that people no longer spend time taking care of rags and scrap metal, blaming it on a more wasteful lifestyle. This perception is somewhat incorrect because the primary reason for not recycling automatically is that waste products lack value relative to current salaries. By contrast, in a poor country such as India, a large fraction of what is thrown away by consumers is recycled.

A major problem with recycling in high-income countries is the lack of demand for recycled goods. In recent years, however, the market for junk materials has been expanding at a considerable rate due to the fast-rising demand for natural resources (Pittel et al. 2010). Almost inevitably, a country with rising income will produce more waste over time, and a government that insists on recycling may have to not only promote recycling but also establish and maintain a market for recycled goods.

One good reason for collecting at least certain categories of waste, even if they have no inherent value, is to avoid polluting the ecosystems where the waste is dumped. However, when too much focus is put on the categories of material that are not hazardous (e.g., glass, scrap metal, and paper), insufficient attention is paid to the hazardous components of the waste stream. Costs for monitoring hazardous waste should be an important component of the analysis; in some cases, waste management fees encourage destructive illegal disposal, so self-enforcing mechanisms like deposit–refund schemes should be considered (see Chapter 7; also see Fullerton and Kinnaman 1995; Jenkins 1993; Kinnaman 2006; and Viscusi et al. 2009).

Some behavioral aspects should also be considered. Households in many countries recycle without direct economic incentives to do so, suggesting nonpecuniary motivations in play, such as moral satisfaction and social approval—an idea supported by several studies (Bruvold et al. 2002; Kipperberg 2007).

In this chapter, we discuss some policies for waste management in different countries, starting with variable waste charges and deposit–refund systems in Sweden and the United States. Next, we move to some developing country issues, then the use of eco-labeling on dishwashing and laundry detergents (which are not waste in the usual sense but affect effluents and water quality). In the final section, we describe the use of a kind of tradable permit for packaging waste in the United Kingdom and the use of a plastic bag levy in Ireland.

Economic Incentives in Waste Management

Differentiated waste fees and other incentives for waste minimization, recycling, and increased composting have been used, particularly in the United States and in some of the more densely populated areas of Europe, such as the Netherlands. Results vary considerably depending on the exact details of the program as well as on local demographic factors (Kinnaman and Fullerton 2000; Repetto et al. 1992; Allers and Hoeben 2010).

United States

Municipalities in the United States, as in most other countries, have traditionally levied fixed collection fees for household waste, or even included them in general (property) taxes. This approach is inefficient because the effective

marginal price of waste disposal is zero, whereas the marginal collection and disposal cost is positive. However, communities are increasingly charging for waste collection based on weight or volume of waste. Such programs have been implemented in more than 4,000 communities (roughly 10% of the U.S. population), and in Minnesota, some form of differentiated disposal fees is mandatory. Programs take different forms: prepaid garbage bags, prepaid stickers to be attached to bags or other waste for collection, collection fees based on the number or size of cans, weight-based systems, or two-part tariffs (a fixed minimum service at a base rate, then marginal charges for excess waste). Two-part tariffs, a relatively recent development, are popular because they are easy to implement, provide a stable source of revenue for collection services, and are perceived as offering a fair previously specified level of service at a fixed cost to most customers.

The effect of these programs has been satisfactory. Among 114 cities with variable rates and 845 communities with traditional fixed rates, the estimated reduction in waste disposal after beginning to charge US\$1 for every 32-gallon bag is 44% (Fullerton and Kinnaman 1995). Other studies have found reductions of 10–75%, depending on circumstances (see U.S. EPA 2001 for a survey). Nevertheless, decreases in waste collection may be misleading if they lead to an increase in illegal waste disposal, which includes dumping, backyard burning, and placing waste in other peoples' containers (including public ones) (see Chapter 7).

Although illegal dumping occurs for various reasons, one of the main factors inducing illegal dumping seems to be a shortage of proper waste management facilities; in the absence of sufficient proper waste management facilities, people are more likely to dispose of their waste illegally to reduce the cost of such disposal (Ichinose and Yamamoto 2011). Thus, the effectiveness of waste collection charges is much higher in cities where recycling programs are run simultaneously. Some features of recycling programs might affect significantly the intensity of household recycling effort. For example, introducing a local drop-off program for recycling of a particular material decreases the time and storage costs with respect to other modes of recycling, such as accumulative material to haul to more distant recycling centers. Curbside collection also increases recycling by eliminating out-of-pocket costs and the need to store recyclables for long periods of time (Jenkins et al. 2003; Sidique et al. 2010).

Sweden

In Sweden, several municipalities charge for waste based on either a per-kilogram or per-bag basis (Sternier and Bartelings 1999; Hage et al. 2009). Furthermore, since 1994 an ordinance has required producers of packaging materials to provide a collection system and households to sort out packaging waste from other waste. Consumers must clean and sort different packaging

materials—paper, plastic, glass, and metal—in assigned recycling bins. Although households' participation is mandatory, in practice, it is easy to defect and free ride on others' contributions. Nevertheless, official statistics show that households in Sweden recycle substantial amounts of packaging materials.

What explains households participation rates in recycling schemes? And how could policymakers encourage further recycling? Several studies have tried to answer these questions, finding that economic incentives, although important, are not the only driving force behind the observed reduction in municipal waste. In a case study in the municipality of Varberg (southwest of Sweden), waste disposal was measured at the household level in Tvååker, a residential area (Sternier and Bartelings 1999). The most important explanatory variables for a household's waste were whether kitchen (and garden) waste were composted, living area (square meters of domicile), age of the residents (recycling efforts increase with age), and the residents' attitudes about the difficulty of recycling. In addition, having in place the proper infrastructure to facilitate recycling motivated people to recycle more than did mere savings on their trash bill.

Moral norms are also important in explaining household recycling efforts in Sweden (Hage et al. 2009): The feeling of moral obligation for recycling, the beliefs about others' behavior, the perceived positive external effects from recycling, and the extent to which recycling generates public goods are all factors that contribute to enhance recycling activities. This means that authorities can stimulate further recycling efforts through information campaigns that emphasize the environmental significance of recycling. Authorities must, however, be careful. Overselling the importance of recycling bulk material such as glass and paper can sometimes obscure the more important issues of avoiding the spread of toxins (e.g., from batteries or printer cartridges) in sensitive ecosystems.

The use of deposit–refund systems has been increasing somewhat for conventional deposit items (e.g., bottles and cans) as well as for such items as scrap cars, where the intention is simply to avoid dumping. Deposit–refund systems are increasingly being used as an economic incentive and an information signal. Ease of deposit, availability of information, and prevailing opinion, values, and habits are important factors affecting recycling behavior. For example, paper is widely collected without any deposit scheme, while the recycling of glass soft drink bottles is an ingrained habit, so recovery rates have been 98% for several decades (Sternier 1999). For aluminum soft drink cans (same contents, sold in the same shops), recovery rates were initially much lower. However, with aggressive marketing and increased refunds, recovery rates rose from about 60% to 90% in the decade after 1984. Similarly, recovery rates for wine and liquor bottles increased from 40% to 70% over a 25-year period before the recycling program of the state liquor monopoly was closed down in the late 1990s. Price elasticities for all these bottles and cans were estimated to be around -0.2 . However, the main result when the deposit increases

seems to be an “attention” effect. When the deposit decreases (in real terms, through inflation), recovery rates do not appear to decline. On the other hand, voluntary paper and glass recycling programs have been successful without any financial incentive; estimated recovery is around 70% (compared with 20% a couple of decades ago). The considerable increase in the number of deposits for these materials indicates that information, social norms, and ease of recycling are important.

For hazardous waste, proper household disposal has become much easier. Large recycling centers have been set up in Gothenburg that accept whatever (hazardous and other) waste the public brings. Furthermore, smaller recycling stations are set up to receive oil, solvents, and car batteries. They generally are located at (and run by) gas stations or small marinas. Passive deposits can be set up for battery collection, for instance, in the streets (close to collection points for glass and paper) or in appropriate shops. Collection vans collect the waste from these deposits and from the recycling stations. The same collection vans also were used for some time for neighborhood service. According to a posted schedule, they visited every neighborhood for hazardous waste pickup.

Passive deposit boxes and small environmental deposits at gas stations have been particularly effective because they fit into people’s everyday routine and thus entail little extra cost. Vans were not cost-efficient because few people used them. Apparently, most people prefer a greater distance at a time of their own choosing to a short distance (or none at all) that requires attention to a schedule.

Waste Management in Developing Countries

Waste management in developing countries is subject to the same general principles as in developed countries, but several important conditions are different. One of these is the abundance of labor with low “opportunity cost,” that is, destitute people who have no other means of employment and thus are prepared to work in some way with the resource that waste represents. Another common characteristic is poor provision of public services (e.g., municipal waste management, water, and electricity). Consequently, in many cities in developing countries, waste is picked through several times after being placed curbside: first by people looking for food or saleable items, then—typically with some delay—by municipal sweepers or waste collectors.

Many other factors make each location unique. In some areas, urban agriculture allows for the recycling of organic matter. In other areas, uncollected garbage presents a significant health risk because of factors such as heat, humidity, insects, and animals. Waste sites themselves are often inhabited by thousands of people who sort through the waste under miserable working conditions. Waste pickers typically sell what they collect to middlemen, who then sell the items to recycling industries. The recycling industry is typically

well developed and may be economically successful, particularly compared with its counterparts in richer countries, where the opportunity cost of labor is much higher and thus the value of recycled items relatively lower (UNDP/World Bank ESMAP 1998).

Because of these differences, the issues and the potential instruments for waste management are different from those in developed countries. Illegal disposal of small amounts of household waste (e.g., on the curbside) is not a major issue; in fact, it is the norm in many places. Waste collection fees may not be the best general instrument (except in more affluent districts). One important issue is the organization of the labor already dedicated to waste management or recycling—a natural target of aid programs. Some have emphasized the role of health services, whereas others emphasize the role of micro enterprises that can use the recyclable items. If scavengers were properly organized, their activities could have a great impact on the economy, and waste management could be considerably improved. An integrated solid waste management approach must be adapted to local social and technical conditions in developing countries (Beukering et al. 1999). Governments must integrate the contributions of existing informal waste collectors, recognizing that their contribution is high. Collaboration with all stakeholders is crucial to integrating the formal and informal recycling agents in Kathmandu, Nepal, where more than 15,000 people make their living scavenging (Bhattarai 2000).

Two topics related to developing countries deserve attention. One is waste production through tourism, discussed in the next section. The other is the dumping of hazardous waste or the siting of hazardous industries in developing countries (see Box 24-1). In principle, this practice is regulated under the Basel Convention, but many unscrupulous firms export hazardous waste under various false classifications (e.g., “fuel,” “fertilizer,” or “raw materials”) to companies in countries that do not have the capacity (or the inclination) to monitor and inspect the materials (see Chapter 21 concerning hazardous wastes and industries in India). All trade is supposed to be registered under the Basel Convention, but the statistics are unsatisfactory. Import statistics for 1998 for non-OECD trade report almost 20 times the quantity that the corresponding export statistics show. Information concerning the treatment of waste (recycling, disposal, or other) is often missing.

Within the convention, work is in progress to formulate agreements on liability and compensation rules as well as to ban the most dangerous of the hazardous pollutants¹. Information is a powerful tool in this context: through the Internet, any journalist—even in a poor country—can access information about hazardous wastes, then contact the press in the country of origin. In the future, this kind of monitoring may turn out to be at least as powerful as (and complementary to) formal legal methods.

BOX 24-1 Hazardous Solid Waste in Vietnam

Vietnam has experienced rapid economic growth during the last decades—7.6% on average for 1996–2005 (World Bank). The establishment of industrial zones has also grown rapidly, with more than 120 concentrated industrial areas generating a large quantity of solid waste, especially hazardous solid waste (HSW) from industrial sectors such as electrical mechanics, food processing, chemicals, and mechanical and metallurgy factories (Thai 2009).

Estimates indicate annual discharges of 28 million tons of waste, expected to increase up to 67 million tons by 2020. Improper disposal of HSW is becoming a serious concern because both treatment facilities that can meet safety and environmental standards and sanctions discouraging illegal disposal are lacking. Unavailable infrastructure is particularly alarming in Ho Chi Minh City—the country’s largest industrial center and one of the most polluted—where disposal capacity meets only about 10% of total needs. Inadequate infrastructure makes the following practices common:

- Hazardous and nonhazardous wastes collected from industries are disposed at domestic landfills without any separation.
- Sludge and toxic liquids are discharged with ordinary wastewater.
- Reuse and recycling are merely informal.

Toxic matter thus permeates underground aquifers, causing water pollution. When dumped into unhygienic landfills, HSW causes negative direct effects on public health. To solve these problems, the Vietnamese government is increasing investment in the construction of central HSW facilities and analyzing measures to encourage the treatment of hazardous waste by private enterprises through licensed treatment services. Initiatives for reducing, reusing, and recycling HSW are being considered. Nevertheless, a number of challenges remain before such policies are implemented in Vietnam because of the increasing quantity and types of waste and the level of the hazard of the waste generated (Thai 2009).

Tourism and Waste Management in the Caribbean

Ecotourism is touted as a benign resource use for sensitive ecosystems². However, the effects of tourism are not always benign; charter boats and most other ships regularly dump wastes, effluents, and other emissions into the very coastal ecosystems that support them. Many of the small “paradise islands” are extremely sensitive—particularly coral islands, which are easily damaged by nutrients, temperature variation, and physical damage. Like many other popular sites, the Caribbean Sea is heavily affected but until recently has had difficulty protecting itself. (Several industrialized countries have started to demand that ships pay for waste disposal at every port of call, but this policy is novel to developing countries.)

At the same time, the management of locally generated waste is a severe problem. In 1995, the World Bank, the European Investment Bank, the Caribbean

Development Bank, and the Global Environment Facility (GEF) joined forces to halt marine deterioration through a solid waste management project in the Organization of Eastern Caribbean States (OECS). The organizers of the waste management project recognize that the ecosystem is both sensitive to waste and valuable (as a source of tourism revenue and for its life support functions); the environmental problem has both local and international causes (making GEF finance an option; see Chapter 14); the generation and dumping of wastes is not always observable, making a Pigovian tax difficult to implement, but some source of local funding is crucial to make the project sustainable (the structure of fees was a fairly contentious issue); and it is difficult to locate suitable land for a landfill site on a small island.

According to the initial appraisal report, the waste management conditions in the OECS—particularly Grenada, the most heavily polluted of the islands—were bad (World Bank 1995). The waste collection service served only half the population and was severely underequipped. Tipping sites were poorly managed, with severe leakage (to aquifers and to the sea). Underground fires and significant odor, rodent, insect, and other health-related problems abounded. In addition, tourists and tourist ships dumped their own waste. The funding agencies were keen to ensure the sustainability of the projects by insisting on a system of fee collection for both domestic and tourist wastes.

To agree to provide finance (US\$50.5 million in a grant and loan package, of which US\$8 million was for Grenada), funding agencies required that the countries improve waste management collection, waste disposal sites, and local waste management and organization, which included implementing fee collection systems. An appropriate tipping fee for domestic wastes would be US\$20/ton, whereas transport and disposal of maritime wastes could be charged US\$40/ton plus US\$35/ton for haulage (World Bank 1995). Furthermore, an environmental fee of US\$1.50 per visitor would be charged. These charges, together with a continued contribution from the public sector and the GEF grant, would be sufficient to pay for waste management services. Notably, the charges were discussed as methods of “cost recovery”; they were not designed primarily to minimize waste (monitoring was considered too difficult).

The construction of a system of fees and other components was carefully balanced to motivate all the parties and legitimize the project. This effort was necessary because there was still a good deal of resistance. Local low-income inhabitants who were used to disposing of trash informally, at no cost, initially were reluctant to pay; gradually, they were convinced of the importance of the program because they saw aesthetic and health improvements and because they perceived that they were not the only ones paying. For the sake of administrative convenience, the fee for waste management was linked to the electricity bill (and related to electricity consumption—far from ideal, but still a workable arrangement).

The largest conflict, which almost stopped the whole project, was with the cruise ship industry. The operators complained bitterly over the “arbitrary”

US\$1.50 environmental fee. In fact, their resistance was surprisingly ferocious, given that this sum was a relatively small share of the cost per passenger, per day. However, the industry lobbied each country, threatening to boycott the countries that imposed the tax and move their business to islands that did not. The World Bank organized meetings between the cruise ship associations and the governments of the region, but no agreement was reached until the countries imposed the fee unilaterally and simultaneously on June 1, 1998. In addition to these fees, Grenada also assesses a 2% waste fee on the importation of vehicles and household appliances as well as US\$0.25 per beverage container. Although these instruments provide much needed finance for waste management, they are not really designed to minimize waste. A partial refund for returning used appliances and so forth has been discussed but not implemented.

The other major area of difficulty encountered by the project was the selection of a suitable disposal site on Grenada. The site initially selected had several good features but turned out to harbor Grenada doves—an almost-extinct species, of which fewer than 100 remain, only on the island of Grenada. This shy bird, which only dedicated ornithologists appear to have noticed, presented a serious threat to the project. Numerous delays and negotiations led to the creation of a dove sanctuary complete with guards, fences, a tourist center—and entrance fees.

By 2000, 95% (rather than 50%) of the local waste was being collected, new safe landfills were in operation, locals and tourists alike were being charged to recover the costs of waste management, and the dove population even showed signs of growth. Whether the cruise ships are only paying the fees or also engaging in any form of source separation and treatment (or dumping in international waters) is not known. The use of fees per ton might encourage both sensible and irresponsible treatment, and such fees might need to be restructured if they lead to perverse behavior. The project also needs to pass the test of sustainability when the donors pull out. According to World Bank project manager Usamah Dabbagh, the project needed an additional year or two of support to guarantee a smooth future (Peck 1999).

This example illustrates the importance for policymaking of a coordinated approach with several components that reinforce each other: collection service; disposal site; fee collection; and cost sharing among locals, tourists, and development agencies. Perhaps the most interesting response was from the cruise ship industry, which was not expected to object to a cleaner environment or to a fee so small that passengers would hardly notice it. The industry is either irrational or, more likely, worried about the precedent set for cooperation or “collusion,” as they see it, among the island states off which it operates. This risk of collusion in other areas may be a by-product of environmental programs and instruments.

Eco-Labeling of Soaps and Detergents

One important waste stream bypasses municipal collection: effluents that drain into sewage. Soaps and detergents—as well as the dirt that they are intended to remove—are one kind of such effluent waste³. Because of the nonpoint-source characteristics of these effluents, it is difficult to imagine any billing or even monitoring system that would detect the amount and characteristics of the waste disposed of through a household's drains. It is therefore natural for policymakers to choose an instrument that applies directly to the product. Standards, rules, liability, taxation, and several other policy instruments could be applied to dishwashing and laundry detergents. The large number of complex ingredients make taxation difficult, and thus the main alternatives are perhaps some form of direct control or voluntary instruments such as information disclosure and labeling. Labeling is also a strategy that may be used in several other waste-related issues, including the environmental regulation of trade.

Laundry detergents have three characteristics that make labeling particularly appropriate: the ecological and technical criteria are complex; the main hazards lie in the product, not the production process; and the product is bought mainly by households (rather than by industries as an intermediary good). The Swedish Society for Nature Conservation has its own eco-labeling scheme, Bra Miljöval ("Good Environmental Choice"), in competition with the more official Nordic eco-labels (see Chapters 8 and 10). In terms of market share, both labels have been successful. The market shares of eco-labeled shampoo, laundry detergent, dishwashing liquid, and sanitary cleaners have risen from 0 in 1990 to 90% in 1997 (50% for Bra Miljöval alone), and the eco-labels appear to enjoy a sound level of acceptance and credibility. If the eco-labeled products really are less environmentally damaging, then this policy will have significantly reduced the toxicity of effluents, decreasing the impact of an important category of household waste.

To become environmentally certified, companies must fulfill a set of criteria and pay a fee. The criteria focus on the ingredients or production process. Proponents of the eco-labeling scheme claim that the companies have been forced to decrease or eliminate the use of various harmful ingredients (such as phosphates) from laundry powders and that this change has resulted in considerable benefits for the environment. This claim was somewhat supported by a study that indicated that phosphates were reduced by more than 50% and perborates by 80% between 1990 and 1995 (Nilsson 1998). However, the criteria contain fairly lengthy lists of ingredients that no laypeople can compare, and the ingredients that are reduced have substitutes, so the use of other ingredients—notably, percarbonates, enzymes, and new kinds of tensides—is increasing. Several other studies report 45% reductions in those components that are the least biodegradable (Rosander 1998; Svenska Naturskyddsföreningen 1999; Wilske 1999) (see Table 24-1).

TABLE 24-1 Reduction of Poorly Biodegradable Ingredients in Swedish Detergents, 1988–1996

<i>Ingredient</i>	<i>Function</i>	<i>Reduction (%)</i>
Ethylene diamine tetraacetic acid	Complexing agent	100
Phosphonate	Complexing agent	50
Pigments	Coloring	100
Fluorescent whitening agents	Optical brighteners	100
Silicones	Defoaming agent	15
Perfume	Fragrance	0

One of the criticisms against the voluntary Type 1 eco-labeling schemes is that they fail to take a life-cycle perspective of the product and fail to take into account the performance and user of the product (see Chapter 8). For instance, making a laundry detergent more environmentally friendly by reducing certain components is good from one perspective. However, if the new formula performs worse or consumers use more of the “green” product (or use it with much hotter water), then the ultimate environmental impact could be much larger than from using the detergent with the old formula. The notion that the eco-labeled laundry detergents are less effective (leading to higher consumption) appears to be incorrect, however, because total demand for laundry detergents in Sweden has fallen by more than 10% from 1989 to 1996—that is, during the period when eco-labeling was introduced, breaking a steady 100% increase from 1958 to 1989.

Another criticism against eco-labeling is that the criteria tend to be static and tend to be renewed almost automatically, without revision, which tends to “lock” the technology at one moment in time and make it impossible to benefit from technical progress (for the good of the environment) (Solyom and Lindfors 1998). Diverging opinions and even conflict concerning the ultimate usefulness of eco-labels is probably inescapable because the natural science, ecological, and technical details are complicated. If not, eco-labeling would not be needed. However, the complexity provides ample room for those motivated by ideology or economic interest to find evidence in favor of their preferred position. Governments and environmental groups—perhaps with little interest in profitability or freedom of trade—can be suspected of desiring to interfere and control. They (at least the environmental groups) also may become dependent on the revenue generated and therefore may act tactically. In fact, governments and environmental groups risk becoming corrupt and viewing the market share of eco-labeled products (as well as the revenues they produce) as a primary objective, perhaps leading to lower standards and a lack of sensitivity to new research on environmental issues that might question the criteria used for eco-labeling. Although no evidence indicates that this is the case, it certainly should be kept in mind as a risk.

Some categories of industry might not want to have independent eco-labeling and therefore fight the principle with profitability arguments. (Industry is divided on this point; some categories of industry argue that the independence

of the certifiers gives the certificate credibility and thus value.) An interesting question is why eco-labeling has been successful in the Nordic countries and hardly anywhere else, with a few niche exceptions⁴. Is environmental consciousness more developed in the Nordic countries than elsewhere? Are ecosystems (or consumers) more sensitive? Are other means of regulation not as available? Or are other means of regulation more available and thus a more credible threat if companies do not comply? Many individual studies would be required to answer these questions fully, and they have yet to be carried out.

At present, we can only offer some clues. One is the oligopolistic structure of retail business in Sweden: three major retail chains, one of which is a consumer cooperative. The environmental organizations appear to have managed to make eco-labeling acceptable and even attractive. The Good Environmental Choice label is a collaboration between the Swedish Society for Nature Conservation and the major retail chains, which is perhaps a practical reason for its relatively rapid success. The existence of a large cooperative may have been important. In soap products, mainstream manufacturers initially tried to ignore eco-labeling in Sweden, but this ignorance led to the emergence and rapid rise in popularity (as measured by market share) of a few small independent producers of supposedly more “ecological” laundry detergents. Because the label had turned out to be so popular with consumers, the large producers were forced to follow suit. However, the same firms continue to resist labels in other markets.

The battle in the rest of Europe has been different from that in Sweden. The large detergent producers are organized in the International Association for Soaps, Detergents and Maintenance Products (AISE), whereas the small green producers (such as Ecover) are organized in the European Association of Environmental Detergent Manufacturers, with only 2%–4% of the market (Nadaï 1999). The result of negotiation between these two organizations was EU detergent criteria that were so lax that they excluded only a few of the worst detergents on the market. Thus, the paradoxical result was that companies didn’t use the label—because it had almost no information in it. The commission had hoped for 10% selectivity (i.e., only 10% of products would be eligible immediately), whereas the AISE wanted 95%—an exclusion of only 5%! This example illustrates the risk inherent in eco-labeling led by industry interests.

Tradable Packaging Waste Recovery Notes

In developed countries, waste management was originally a freely provided public service⁵. The need to harness economic incentives in this area has led to a good deal of experimentation with deposit–refund system (DRS) and waste management charges. Several countries also have introduced landfill taxes and other policies.

The United Kingdom has a policy that appears to be the latest and most sophisticated in the sense of harnessing the efficiency of the market in the

quest for materials recycling. The background to this policy instrument is the EU Packaging and Packaging Waste Directive (94/62), which was transposed into U.K. law by the so-called Packaging Regulations, which set annual targets for each year up to 2001. It required the United Kingdom to meet targets for the recovery and recycling of packaging waste (set as a percentage of total packaging) by the end of 2001: 50% for total packaging with a minimum 25% recycling and an individual minimum 15% recycling for each material (e.g., plastic, steel, or paper).

Alongside, the United Kingdom introduced the Producer Responsibility Obligations Regulations in 1997 (amended in 1999) to meet these targets. Unlike most EU member states, where recycling targets are focused on manufacturers and retailers, responsibility in the United Kingdom is shared by all companies that produce or supply packaging. Thus, responsibilities were divided between raw-material producers, packaging manufacturers, product manufacturers and retailers, though the last two sectors shoulder the greater share of the burden.

In designing a strategy to meet the Waste Directive, the British government attempted to fulfill three main objectives. First, a system was needed to monitor the recovery and recycling of packaging waste by producers. Second, the system should incorporate the polluter-pays principle, but avoid unnecessary disruption of business activities and competitiveness. Finally, the system should provide a source of funding for major expansions in the British recycling industry, which at that time was one of the less developed in Europe (O'Doherty et al. 2003). The choice was the Packaging Recovery Note (PRN) system. Under this regulation packaging producers can pass their recovery-recycling obligations on to the scheme, which takes on this obligation in return for membership fees based on the producer's original percentage recovery and recycling targets. Accredited reprocessors receive packaging waste and issue completed PRNs that are used for compliance. By allowing reprocessors to charge for PRNs, this system provides funds for recycling activities and infrastructure. The UK experience with the PRN system opens up other more efficient methods that decisionmakers may not even have imagined. Several features of this system—encouraging technical and organizational progress, cost sharing, and including all the involved parties—are desirable attributes in any policy instrument.

The Irish Plastic Bags Levy

In 2002, Ireland took the lead to reduce the consumption and improper disposal of plastic bags by introducing a €0.15 tax on plastic shopping bags previously provided free of charge to consumers at points of sale. This tax was designed to change consumer behavior toward more sustainable shopping bags, reduce the presence of plastic bags in the rural landscape, and increase public awareness of littering (Convery et al. 2007). Indeed, the tax is so high—more than six times the average maximum willingness to pay for a plastic

bag—that consumers prefer to bring their own permanent reusable shopping bags with them instead of paying the tax.

To ensure stakeholder and consumer acceptance of the tax—from a political point of view, quite important given the levy’s magnitude—a strong publicity campaign by the Department of Environment, Heritage, and Local Government was carried out to convey the reasoning behind the levy and the hypothecation of revenues into an environmental fund.

The effect of the tax has been dramatic, inducing a reduction in use on the order of 94%. The combined effect of the public campaign and the fee seemed to have done the trick. In countries where the customer is charged, say, 15 cents by the shop for plastic bags, little effect is seen on demand. In this case Ireland’s minister for the environment specifically targeted consumers with a combination of a sudden fee and a publicity campaign, thereby changing behavior drastically. The presence of plastic bags in the landscape has also decreased significantly. Litter surveys have shown that after the introduction of the tax, “clear areas”—areas free of plastic bag litter—have increased by 21%. The number of areas with only “traces” (up to five items over a linear distance of 1 meter) has increased by 56% (Convery et al. 2007). These numbers are quite impressive given the long-lasting nature of plastic bags in the environment. Costs of administration have been very low since the reporting and collection of the tax is integrated into the existing Value Added Tax reporting systems. Unsurprisingly, many countries and states are planning or have implemented similar proposals (as for example, the UK, China, Kenya, and the District of Columbia in United States). The issues and policies needed in each country are, however, likely to vary significantly.

Two main lessons might be learned from this experience. First, the introduction of a strong price signal can influence consumer behavior significantly. Second, social acceptance of the tax is a key to successful implementation. When Convery et al. (2007) conducted a series of interviews with households and leaders in the retail sector to evaluate their attitudes towards the tax, they found a common perception that the tax has been positive. Some of the households surveyed stated that they even feel guilty when they forget to bring their own long-life bag and have to pay the levy. Needless to say, it is uncommon for taxes to induce such enthusiasm and affection from those paying them. Extensive consultation with stakeholders to identify key constraints before launching the program seems to have been a critical issue for its wide acceptance.

Supplemental Reading

Beukering 2001
Hage et al. 2009
Kinnaman and Fullerton 2000
Kinnaman 2006

Palmer, Sigman, and Walls 1997
Pittel et al. 2010
Viscusi et al. 2009
Watabe 1992

Notes

- 1 The Ad Hoc Working Group of Legal and Technical Experts To Consider and Develop a Draft Protocol on Liability and Compensation for Damage Resulting from Transboundary Movements of Hazardous Wastes and Their Disposal met for a tenth session in Geneva, Switzerland, during August and September 1999. The aim was to set international rules regarding liability and compensation for damage resulting from the transboundary movements of hazardous wastes and their disposal (see UNEP 1999).
- 2 Thanks to Phil Hazelton, John Dixon, and Julia Peck at the World Bank for information on this project.
- 3 There is some substitutability between conventional household waste and sewage. In the United States, garbage disposals (which chop up kitchen waste and feed it into the sewer) are common. They effectively convert solid waste into liquid effluent.
- 4 Organically grown foods have filled such a niche in many countries. “Green” electricity and “dolphin-free” tuna are two of the few U.S. examples. Indonesia’s PROPER is one of the few developing-country examples (see Chapter 21).
- 5 Thanks to Frank Convery for information about and discussion on this instrument.

Fisheries

Fish are an important source of high-quality food, particularly for the poor, for whom small quantities often provide the only protein in a diet based on a staple food such as rice. Fishing is an important source of employment for many coastal people¹. The commercial value of fish presumably far outstrips the value of any other wild game that humans hunt. Fishing also represents a considerable recreational value. Finally, the stock of fish is a vital indicator of the state of the environment; fish live in the water that covers most of Earth's surface and also is the final deposit for a large share of industrial and agricultural pollution.

Cultivation of fish has increased significantly, and some people foresee a future in which wild fish will be displaced by aquaculture, as has occurred with terrestrial ecosystems. Although this possibility remains, it would pose some ecological and social problems. World production of fish and crustaceans reached more than 152 million tons in 2006 (Figure 25-1). Aquaculture already accounts for more than one third of this production and an even larger share of the production directly used for human consumption. If current trends hold, cultivated fish will be on a par with captured wild fish for the first time in history by the time this book reaches its first readers!

Fish farming as a complement to agriculture is an old tradition, and ten years ago the FAO wrote that it ought to be modernized, copied, and expanded (FAO 2002). This has already happened. The oldest known references are from China (4,000 years ago), and China still takes the lead internationally in fish farming. For instance, in 2006 China accounted for more than 30% of this production. Integrated farming systems could provide not only nutritional advantages but also income advantages by reducing risk (through income diversification), thus increasing harvests and profits. They also could provide associated environmental benefits—such as the biological control of mosquitoes, other disease vectors, and agricultural pests without the use of insecticides or pesticides—and complementary advantages because the ponds can be sources of both fertilizer and irrigation (FAO 2000).

To understand fisheries, several major kinds of fishes and fisheries must be distinguished as well as the complex ecosystems that they are a part of. On the one hand, demersal fish, which feed on continental shelf and ocean

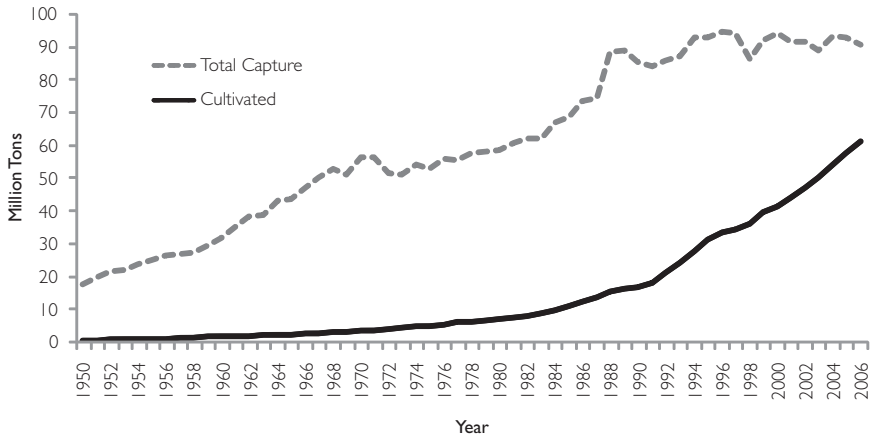


FIGURE 25-1 Wild Harvest of Fish, Crustaceans, and Mollusks Worldwide, 1950–2006

Source: FAO Statistics 2009

bottoms, are particularly abundant in shallow waters and typically are a main source of local subsistence fishing in developing countries. (The same is partly true of freshwater fish, aquaculture, and some nonfish species such as squid and crustaceans.) On the other hand, pelagic fish such as tuna, which live in deep water, are caught mainly by large commercial fishing fleets. There is a good deal of overlap—and conflict—between the various kinds of fishing.

Many of the major fish stocks appear to be harvested at (or beyond) their maximum sustainable yield. Prior to the late 1980s, catches of many of the economically important species increased rapidly for several decades. However, for most fisheries (including many of the most important commercial ones), these increases have been followed by stagnation or decline. In general, stagnation has not been due to less fishing effort. On the contrary, fishing fleets have grown in size, number, and technology. The largest trawling ships are enormous industries with nets so large that several jumbo jets would fit inside, and at the same time, these ships have all the best sonar, global positioning system (GPS), and other technology to find the schools. The industry rarely identifies an appropriate level and sticks to it. Time and again, overfishing—often despite clear warnings from fishery biologists—has resulted in crashes in fish stocks and numerous conflicts between fishing fleets. In some cases, these crises have led to postcrisis management schemes that may allow the fish stocks to recover enough to be fished sustainably. From 1999 data show an increase, again discontinuing the dip in 1998. However, these world fishing data have recently been severely questioned by experts who argue that they are seriously misleading for two reasons: temporary effects related to El Niño and, more important, erroneous data from China, a leading fishing nation. According to Watson and Pauly (2001), the Chinese reported increases in capture fisheries that were so large Watson and Pauly do not believe they are

possible. Strikingly, there would be a very strong incentive for everyone today to stop underreporting and maybe even over-report in case they believed that world fisheries will soon be regulated by treaties based on grandfathering with respect to historic catch figures. This would certainly be analogous to what has been observed in the area of carbon emissions.

In fisheries, “the market” generally fails without regulation for the simple reason that the basic requisite of resource ownership is not present (see Chapter 22). Thus, a policy to avoid overfishing is needed. However, the market failure usually is supplemented by policy failure, because the policies in place have the opposite effect. Instead of restricting effort, helping prevent overuse, and allowing stocks to recover, many policymakers simply “help the fishermen,” who are perceived as disadvantaged and vulnerable. Thus, credits and subsidies are provided for faster and bigger boats, longer and more finely meshed nets, and new fish-finding equipment. All these technologies lower the cost of fishing but expand capacity beyond the optimal—sometimes even beyond the sustainable—level, making overfishing worse. Subsidies are—or at least were—an important element in lowering production costs and expanding fisheries. In more recent OECD estimates, subsidies appear to be lower; OECD subsidies in 1997 were calculated at US\$6.3 billion, which is still sizable. The costs of fishery management in Iceland, Norway, and Newfoundland (Canada) are considerably different, indicating differences in efficiency as well as the distribution of costs between fishermen and society.

Subsidies that lower the cost of fishing are bad policy if the main problem is overfishing as a result of open access. However, several policies appear to work, including partial property rights (such as the 200-mile exclusive zone), fishing licenses, restrictions (e.g., on gear or seasons), and fishing quotas. The most direct policy instrument is to set catch limits in the form of a total allowable catch based on maximum or optimum sustainable yields; however, they must be enforced, for example, through individual transferable quotas (ITQs). Other possible instruments would be taxes and information or voluntary instruments such as labeling schemes.

Restricting access is a popular instrument because it targets “outside” fishermen—from other countries, regions, or villages. This mechanism is fundamental at the local level of common property resource (CPR) management and also is the main purpose of the enclosure of the seas implicit in the extended fisheries jurisdiction (see Chapter 3), under which 90% of the world’s fish catch is based on resources that lie in nationally controlled waters (Clark 1990). It will not necessarily lead to sustainable management but can be seen as a prerequisite to other instruments at the national or local level, such as CPR management or ITQs.

When the catch is too difficult to monitor accurately, technology might appear to be easier to regulate. The prohibition of destructive technologies such as dynamite and cyanide is a positive advancement. However, many other technologies are banned simply because they are too effective, even though

they are not strictly harmful². Fishermen who use simple, small-scale technology often perceive the more “advanced” equipment as unfair and lobby against it. Technology restrictions are common and, in some cases, effective in protecting stocks—but at the cost of depleting the rent that should be protected. A better approach would be to use the new technology but reduce effort and catches to an optimal and sustainable level (see Chapter 22). Regulating technology would mean monitoring each and every detail of fishing capacity (Brown 2000). If the number of boats is limited, then owners acquire bigger boats. When regulators fix the maximum length of boats, boats become rounder and more powerful. In areas with different length restrictions, boats have detachable “auxiliary” bows.

Restricting fishing seasons, another indirect method used when catch is difficult to monitor, also has several disadvantages. Resource depletion is not the only problem fisheries face—although it tends to be a dominant problem. Other important issues relate to occupational safety and to the value of the fish, which is strongly related to its freshness. Certain restrictions (particularly fishing seasons but also total physical quotas) tend to encourage derby-style fishing during the short period the fishery is “open.” Some halibut fishermen adapted to a two-day season by investing in three identical electronic systems, where two were backups. This extreme example shows how restricted fishing seasons can lead to what is referred to as “overcapitalization” or “capital stuffing” (Homans and Wilen 1997). The extra capacity may be idle part of the year or migrate to other areas, contributing to overfishing elsewhere. Short seasons also lead to lower commercial value of catch, because it has to be sold frozen most of the year. Finally, it leads to fishing even in bad weather, which increases safety risks (NRC 1999).

A moral hazard issue can result from the link between fisheries management and safety (Bergland and Pedersen 1997). The authority (the principal) chooses public supply of effort on safety, then the fishing operation vessels (the agents) decide on levels of private inputs. Public engagement in safety issues is necessary to supply public goods such as communication and navigation systems used by all vessels. The moral hazard problem arises because the supply of public services might encourage individual fishermen to behave in a way that increases risk.

A tax would be an excellent solution if the focus were only on the allocative effects, but a tax is politically hopeless because the government would then remove the rents from the fishermen it purports to protect (see Chapter 22). Refunding taxes would not be an option because the level of fishing, not the technology of fishing, is being regulated (see Chapter 7). Refunded taxes will not have the “output effect” of a tax (see Chapter 7 and Chapter 11). If refunding were tied to output, then refunded permits would not be a reasonable way to reduce overfishing. However, a tax on catches, the revenue from which would be used for community development (e.g., infrastructure, aquaculture, tourism, or other public goods), might be an option³.

The appropriate instrument should also allow for some flexibility in this level as stock assessments vary. Although common assessment techniques and management strategies assume discrete populations (see Chapter 22), stocks are complex: multiple populations and subpopulations play an important role in ensuring stock viability and genetic variability. Persistence of diversity of stocks should become a principle of management, particularly under a “precautionary approach,” since fish of different subpopulations do not breed with each other, a factor which should be taken into account in models estimating allowable catches. Svedäng et al. (2010) show how cod subpopulations in Scandinavia may have been eradicated as a consequence of using imperfect models for assessing available fish resources, putting a formerly productive sea in a steadily depleted state. The study indicates that policy instruments are needed, but these instruments need to be very carefully fine-tuned to take into account real biological as well as social factors.

In this chapter, we focus on two instruments: CPR management and ITQs. The former appears to be more appropriate for small, coastal fisheries and the latter for large commercial fisheries; however, this does not mean that policies such as fishing licenses and technology requirements are always inappropriate.

Management of Fisheries in Developing Countries

For many developing countries fish is an important source of protein, employment, and exports. While trade is generally beneficial for growth and welfare, the combination of open access and trade liberalization may reduce both welfare and stock of a country (Eggert and Greker 2009). Subsistence fishing is also difficult to manage centrally because of the highly scattered nature of landings, transport difficulties, and the difficulty of enforcing regulations in remote areas (Baines 1982; Nietschmann 1984). Traditionally, these fisheries have been managed locally, not centrally, and they provide many good examples of CPR management regimes (see Chapter 8). Mechanisms for excluding nonparticipants are important for CPRs, and access to sea resources may be limited by three means: restricting information (e.g., fish location and suitable methods of fishing), actively defending fishing sites (by sabotaging rival gear), and formal exclusion (Durrenberger and Palsson 1987).

Community-Based Systems

In traditional management systems, it is common to find the entire community involved in managing fishing resources. This kind of system has been documented in the Indo-Pacific region, the Caribbean, Latin America, and Africa, as well as in many industrialized countries (Ruddle, Hviding, and Johannes 1992). In Japan, feudal rulers historically possessed fishing rights, which they gave to villages in exchange for taxes (see Chapter 3). Today, local fishing cooperatives control many of these fishing grounds (Akimichi 1984).

In the Pacific Islands, customary marine tenure systems are common. Membership in kinship groups includes user rights for both land and sea resources. Membership may be based on ancestry or obtained through marriage or residence, depending on the region, and complex rules sometimes govern user rights (Hviding and Baines 1994). Fishing rights may correspond to specified areas inside a barrier reef or another important fishing ground. Within these areas, outsiders must obtain permission to fish. Coastal residents may exchange fishing rights for farming rights with groups that live inland. Coast-dwelling clans may negotiate if a certain area is lacking in some species or if they experience inclement fishing conditions. Generally, outsiders will be allowed to fish for subsistence but not for commercial purposes. Access to items that have only commercial value (e.g., pearls, coral) usually is severely restricted, enforced through social pressure, and violators are publicly disgraced. Compensation may also be exacted for illegal fishing. Resource owners may contact the local police to prosecute illegal fishermen, and habitual poaching or trespassing may result in social excommunication.

Traditional management centers on control of effort by controlling methods, location, and timing. Fishing gear and net size are commonly limited, explosives are almost always prohibited, and the use of poisonous plants and underwater spear guns is typically restricted. Certain species are not to be fished—such as eels, turtles, and crocodiles—or it may be taboo to fish a certain species at a certain time, such as juvenile fish in shallow waters. Seasonal closures on some or all fishing grounds are designed to allow stocks to recover. Certain areas may always be closed to fishing. Fishing sites are often rotated following cycles of the moon, tides, or wind, thus preventing overfishing in any one site. Rotation is a good method to distribute heterogeneous grounds fairly and also makes it easy for fishermen to monitor each other and enforce regulations at little extra cost as part of their own fishing activities (see a description of the Alanya fishery in Turkey in Ostrom 1990). Generally, the rules concerning allowable species and timing adapt to changing ecological conditions and to the needs of the community. The managers and fishermen have extremely detailed knowledge of the ecology of their systems; in one area, more than 60 fishing methods, 40 reef features, and 400 fish types are specified as well as the migration paths and spawning behavior of various fish.

This traditional kind of fisheries management is sometimes known as territorial use rights fishing (TURF). In the Caribbean, one of the few documented cases of TURF is in Jamaica, where fishing management within a reef area resembles that in the Pacific Islands. Social control of access to certain territories restricts fishing. Enforcement takes the form of interference with fishing traps by the controlling group. This kind of social management normally would not be expected to be efficient in the sense of maximizing rents, but it performs better than open access in this case; Jamaica had higher fishing yields than the average Caribbean fishery (Berkes and Shaw 1986). TURF management also may provide incentives for marine farming as well

as marine parks and ecotourism: marine aquaculture has been introduced into TURF areas in numerous islands (Schug 1996), native groups in Panama and Papua New Guinea have requested that their traditional territories be made into national parks (Nietschmann 1984), and parts of Mafia Island (off Tanzania) are becoming a marine park (Andersson 1995; see Chapter 28 for more examples).

Entry of Commercial Fishermen

In many countries, an influx of commercial fishermen into traditional fisheries has created hybrid management systems in which the CPR institutions tend to break down (Cordell 1984; Ruddle 1988). Often commercial and subsistence fishermen directly compete for resources. Commercial fishing can be a positive force, bringing new technology and higher prices; however, in the absence of clear property rights, commercial fishing has destroyed some fisheries in North Yemen, East Java, Thailand, and India (Thomson 1980). In Malaysia, powerful boats and gear used by commercial fishermen are destructive to traditional fishermen (Vincent and Ali 2005). Commercial fishing creates pollution, degrades habitat, encourages overexploitation, and interrupts the fish food chain. In addition, heads of clans who sell resource rights to commercial fishermen and do not distribute the proceeds to everyone in the group have created social inequality in local communities.

Traditional systems may break down in response to many changes, for example, the effects of money transactions on previously non-profit-oriented fisheries, the loss of authority of traditional rulers, and the enforcement of new laws by colonists (Johannes 1978). Overfishing after commercializing previously subsistence fisheries has been reported for salmon in British Columbia (Rogers 1979) and for turtles in Nicaragua (Nietschmann 1972). Population growth is thought to play a role in the degradation of traditional management systems in Nicaragua (Nietschmann 1972) and in Micronesia (Johannes 1978). Growing populations in Eskimo and First Nation communities in northern Canada increase pressure on natural resources; government-sponsored resettlement of a scattered and nomadic population in the James Bay Cree fishery had similar effects. Resources need to be appropriated either through granting monopoly rights or quotas, in the case of commercial fisheries, or by giving small-scale fisheries control over their fishing grounds (Berkes 1985).

Opening up export markets and allowing the entry of foreign processors may be destabilizing to CPR management systems. Effort and capitalization increase, and fishermen borrow money to invest in larger crafts and more expensive gear. Even fishermen who realize that the fish stock is being depleted may continue to fish because they need to pay off their debts. Once the traditional systems have broken down, they are difficult to rebuild, because the traditional leaders have lost authority and because fishermen who have become accustomed to having free access resist attempts to limit their effort

(Johannes 1978). In the island of Ouvea, one of the Loyalty Islands off the coast of New Caledonia, traditional authorities maintained their control even after the introduction of modern technology; when the sites were overfished, the tribe reverted back to its traditional conservation practices (Nietschmann 1984). Several traditional fishing communities have written constitutions to preserve and make explicit allowances for traditional fisheries management.

On Zanzibar Island, Tanzania, fishing effort has exceeded sustainable levels for several years (Mkenda 2001). Its fishing villages appear to have had a fairly effective CPR management originally, but various factors led to a rapid increase in the number of fishermen and vessels: the decline of the agricultural economy, subsequent unemployment, and massive entry into subsistence fishing, as well as the entry of large commercial vessels. Some local decisionmakers appeared to believe that catches could be increased infinitely by increased effort.

Effect of New Technology

Technology imports specifically aimed at increasing the living standard of native fishermen may be counterproductive. If fish stocks are near their maximum sustainable yield as a result of population pressure, then technology improvements may further erode the stock and create income inequalities, because certain fishermen can afford new gear and others cannot. The least efficient fishermen may be driven out of the market as fishermen with better gear deplete the scarce supply. In San Miguel Bay, Philippines, the entry of trawlers in 1980–1982 caused a redistribution of income. In an effort to increase the dwindling catches of small-scale fishermen, the government subsidized motorized boats and nylon netting, exacerbating the overfishing problem. Meanwhile, the overall numbers of fishermen continued to increase, despite declining catches, due to poor employment opportunities elsewhere (Baines 1982). The Malaysian government supported traditional fishermen in the 1960s and 1970s in their competition with larger vessels by heavily subsidizing the traditional sector. When overexploitation resulted, the government implemented a restricted-entry licensing program for inshore waters in the 1980s and cut back its subsidy program. However, because large fishing firms had more political power than the traditional sector did, violations were seldom punished (Vincent and Ali 2005).

Sakthi, a traditional fishing village in Kerala, India, was the site of a fishery development project in the 1950s. Some reported beneficial aspects of modernization include increased specialization and productivity, risk reduction as catch variance is lowered, and reduced effort. Motorized boats allow offshore fishing, breakwaters allow for fishing in rough weather, and storage facilities smooth price fluctuations. However, the negative effects include social stratification and loss of social cohesion. Owners of new mechanized boats were sometimes unable to pay for maintenance and repairs and lost

ownership. Around the same time, the lucrative export market for crustaceans was emerging. Large investors such as Union Carbide, Tata, and Kelvinator entered the crustacean export business, and total catch increased dramatically. Soon, fish prices increased beyond the means of local residents. Previous small boat owners became crew members on the trawlers, and the line between crew member and boat owner became increasingly difficult to cross. The traditional sector decreased steadily in size; by 1978 there were one ninth as many traditional boats as in 1953. The mechanized fishing sector experienced an inflow of workers from other industries, and land prices near fishing locations increased. More recently, signs of prawn overfishing have been apparent, in which case both boat crews and traditional fishermen will suffer (Platteau 1984).

Sometimes the asymmetries of information and the difficulty or cost of monitoring may lead fishermen to adopt input regulation. For example, a common strategy for limiting the total annual catch in a fishery is to restrict entry and season length. Fishermen have considerable difficulty on their own building up the institutions to limit harvest and thus protect stock, and as a result, they may resort to input control—especially if they can earn some rents due to the restricted capacity (Scott 1993; Hicks and Schnier 2010; Deacon et al. 2011). In addition to information asymmetries, the difficulty of dealing with heterogeneity, entry, and distributional issues are key factors. Ecological reasoning—such as the importance of age and size for recruitment—argues in favor of marine reserve areas or zoning as a prime instrument (Pauly 1997; see also Chapters 22 and 28). If ecosystems are very complex, it may be preferable to leave one zone fairly untouched as a source of larvae and other ecosystem “services” as well as a gene bank and a form of insurance in case the management of commercial areas is unsuccessful.

ITQs in Fishery Management: A Review of International Experiences

Since the late 1970s, when countries began to enclose the ocean commons by establishing exclusive economic zones, several countries followed New Zealand’s and Iceland’s lead in establishing individual transferable quotas (ITQs) to manage fisheries. Fisheries appear to have several special characteristics that make ITQs particularly effective: the high value of the resource, the mobility of the resource (and thus the difficulty of creating ordinary “private property rights” based on territory), and the strong negative externalities exercised by one fisherman vis-à-vis the others (Christy 1973). ITQs have become accepted and are, with some cautious reservations, endorsed by the National Research Council’s Committee To Review Individual Fishing Quotas (NRC 1999). By 2005, ITQs were used to manage over 80 species (Newell et al. 2005). Additional characteristics of ITQ programs are listed in Table 25-1, which summarizes how different programs dealt with the tricky questions of allocation of rights and monitoring, excessive fishing effort, overcapitalization,

efficiency, administrative and enforcement issues, and issues such as safety and fish quality.

Iceland is a particularly interesting example of the use of ITQs because it is among the larger fishing nations in the world; fishing is such a dominant industry—its direct share in gross domestic product is more than 15%—that policymakers take it very seriously. In the 1960s, serious conflicts (the so-called fishing wars) broke out between Iceland and the United Kingdom, eventually becoming an important factor in the formation of the Law of the Sea. Icelandic fisheries also have experienced many of the problems typically found in other countries' fisheries. The value of fishing capital in Icelandic fisheries increased by more than 1,200% from 1945 to 1983 while real catch value increased by only 300%, indicating that overcapitalization (and overexploitation in general) was a serious problem. Effort restrictions, moratoria, and overall quotas did not prevent fish stocks from being reduced to a drastically low level, at which point the fishing industry began to cooperate with management plans. In 1979, Iceland was the first country to implement an ITQ system, initially for herring. The program was expanded several times to include more species until 1990, when a system that encompassed all species in Iceland was created.

ITQs in Iceland were superimposed on the earlier management system; thus, a policy of limited access based on nontransferable licenses still exists. Initially, in the capelin (a small smeltlike sea fish) fishery, 33% of quota shares were allocated on the basis of vessel capacity and the remainder were divided equally; in the herring and shrimp fisheries, shares were divided equally. The Ministry of Fisheries sets the total allowable catch (TAC). TACs for certain species (such as cod) have been set above the biologically recommended limit—allegedly “because of the dependence of the Icelandic economy on fishing.” Quota trading has been strong: since 1986, 20%–30% of quotas have been exchanged annually; in 1993–1994, 44% and 96% of quotas were traded for cod and saithe (the Atlantic pollock), respectively.

Ample evidence demonstrates the economic benefits ITQ in Iceland has yielded (Arnason 2008). New investment in fishing capital has been reduced, and the fishing fleet has contracted. In some fisheries, ITQ has also succeeded in restoring stocks. Nevertheless, ITQs appear to have increased the wasteful practices of disposing of immature fish and discarding less valuable fish when better fish are caught (highgrading) in multispecies fisheries. Although the quota system was originally presented as a temporary emergency measure, it has caused a major restructuring of the fishing industry, highly concentrating it and leading to some fierce distributional conflicts. In addition, the ITQ-values of the Icelandic fisheries have risen quite dramatically. In 1984, the ITQ-value was about US\$25 million. Since 1998, it has hovered between US\$3.5 billion and US\$4.5 billion (Arnason 2008).

The experiences of other countries vary, but overall they corroborate many of the Icelandic experiences, showing that ITQs can succeed in reducing

TABLE 25-1 Characteristics of Some ITQ Fisheries

Country: Area	Year	Species	ITQ alloca- tion	Cost	Basis	Property rights ^a	Enforcement	Payment	Economic results
New Zealand	1986	32 species	F	Initially free	H	Perpetual, full rights (except rock lobster) ($R_{\max}$)	Auditing	Violation is a criminal offense	Output 0 Employment 0 Fish quality + Quota price +
Iceland	1979 1984 1990	Herring Demersal All	V	Free	H/C/E	Annual revocable vessel quota (R_{conc} , R_{empl})	Auditing	Costs paid by industry	Catches +/0 Effort – Catch quality + Profits +
Canada	1991	Halibut	V	Free	70% H 30% C	(R_{conc})	Catch monitoring	Penalties: forfeiture of quotas; informers get shares	Catch 0 Prices + Employment – Concentration restricted
Chile	1992	Red shrimp and cod	F	Auction		($R_{\max}$); valid 10 years; annual auction of 10%	Weak; self-monitoring	Graduated fines	Rents increased
Norway	1973	Herring, mackerel, blue whiting, capelin	V	Free	C	Restricted transfer subject to approval by Ministry of Fish			
Australia	1984	Southern bluefin tuna	F	Free	75% H 25% C	Freely tradable	Catch monitoring	Cost paid by industry (44%)	Catch/effort + Rents + Capital used –

TABLE 25-1 (Cont.)

Country: Area	Year	Species	ITQ alloca- tion	Cost	Basis	Property rights ^a	Enforcement	Payment	Economic results
United States	1992	Wreckfish	V	Free	50% H 50% E	(R_{conc}); perpetual but not full property right	Monitoring by "catch coupon," landings inspection	Violation leads to fines	Overcapitalization – Efficiency + Effort – Price stabilized
East Coast	1990	Surf clam and ocean quahog	V	Free	80% H 20% C	Quotas full property	Shoreside surveil- lance, cage tags, logbooks		Efficiency + Catch/ vessel, rent + Excess capacity – Economic outcomes uncertain
Alaska	1995	Halibut and sablefish	V	Free	H	(R_{conc})	Accounting and auditing (debit cards, fish tickets)		
Florida	1992	Spiny lobster	F	Fixed price	H	(R_{conc})	Monitoring of trap tags; inadequate		Number of traps – Landings stable Value of permit +

Notes: ITQ allocation is to fishermen (F) or to vessels (V), and ITQ basis is historic (H), capacity (C), or equal (E). ^a“Property rights” generally does not mean full property rights, but freely tradable except for retrictions as noted. Restrictions to free tradability or to property rights are based on share of fish or permit holdings (R_{max}), concentration (R_{conc}), or employment effects (R_{empl}).

Sources: Arnason 1996; Batkin 1996; Casey et al. 1995; Flaaten, Heen, and Salvanes 1995; Gauvin, Ward, and Burgess 1994; Geen and Nayar 1988; NRC 1999; Wang 1995.

the risk of stock depletion (Costello et al. 2008). Currently, ITQs are generally a proportion of the TAC. However, in New Zealand, shares were originally specified in terms of fish tonnage. The initial allocations—based on past catch histories—exceeded the maximum sustainable yield in some fisheries, so the government had to buy back quotas on two occasions. These quotas turned out to be very expensive.

Allocation is usually to individual fishermen (e.g., in New Zealand and Chile and for spiny lobsters in Florida) or vessel owners (e.g., in Iceland, Norway, and some U.S. fisheries and for halibut in Canada). Allocation to individual fishermen is more suited to encouraging small-scale operation and in some contexts might better satisfy distributional concerns.

The basis for allocation defines the character of each program. Some programs have a grandfathering approach whereby rights are based on historical catch. Other systems base rights on vessel size or capacity; commonly at the inception of programs, small boats are not included in the ITQ system and may catch as much as they like, although eventually (especially if there is a large noncommercial subsistence fishing sector), small boats must be regulated. Shares are sometimes allocated equally among all participating boats. Systems often use more than one of the above criteria for determining rights, but shares are almost always allocated free of charge. One exception is Chile, where rights are auctioned within the ITQ system; however, only two minor fisheries are involved in the Chilean ITQ system. In many countries, auctioned permits, like taxes, would generate considerable resistance. However, partial auctioning or allocating permits with some intermediate price might be an acceptable option.

New Zealand has (at least partially) defused the potential conflict between traditional and commercial fishermen. Commercial fishermen harvest their share of the TAC after tribal and recreational fishermen have fished. When the TAC is increased, commercial fishermen must purchase additional quota shares, whereas tribal and recreational fishermen have free access (subject to conservation laws), so social dislocation has been minimal. Another example of minimal dislocation is in the U.S. wreckfish fishery, which introduced ITQs without much economic dislocation, mainly because the fishery is relatively new: quotas allocated on the basis of catch history did not have a significant redistributive effect. In the U.S. halibut and sablefish fisheries, part of the TAC is allocated to six nonprofit communities as part of a community development program. Concerns over community development were one impetus in introducing ITQs. The program does not apply to subsistence, treaty, or sport fishermen. Still, some dissatisfaction exists as to the initial quota allocation, which excluded crew members and processors.

The nature of the ITQ varies from being a full legal property right (as in several cases in New Zealand), to being “perpetual” but not a full property right (as in the U.S. wreckfish fishery), to being nominally the property of the government (as in Iceland). However, the high cost at which permanent quota

shares are traded in Iceland suggests that quotas are considered personal property that, although not private, is not likely to be revoked by the government.

In most cases, shares are permanent or of indefinite duration, and this certainty is considered an important aspect; in at least one instance, authorities have distributed permits and then been forced to revoke them in a rock lobster fishery in New Zealand (Breen and Kendrick 1997). The exception is again Chile, where quotas are valid for 10 years. Quotas are sometimes subject to restrictions on trading, usually to protect share ownership or local employment in certain communities. Such is the case in the Canadian halibut fishery, where the ITQ program was initially a two-year experimental program that did not allow trading. In 1993, limited trading was allowed; currently, vessels cannot trade more than twice their initial allocation of shares and can lease shares only annually. Transfer restrictions are intended to limit consolidation of the fleet.

Taxation is an alternative policy that could entirely replace ITQs. In a model with uncertainty in the assessment of stock and recruitment, the most essential variable is the size of the surviving stock, which depends on the unknown (and stochastic) recruitment of new fishes. The inherent flaw of quantity regulation is that the size of the TAC and the quotas must be determined before the regulator knows these variables and thus the appropriate catch. The correct tax may be easier to know if, as is commonly assumed, the cost of fishing varies with the density of fish. If the target stock is a certain minimum level, then all the tax has to do is remove the profitability of any fishing effort that would bring the stock below that level. Although such a tax scheme sounds indirect and complicated, a model with inherent ecological uncertainty that clearly favors taxes over ITQs from the viewpoint of efficiency has been developed (Weitzman 2002).

Enforcement and Economics

Successful monitoring and enforcement are essential for any fishery management regime. Nevertheless, monitoring and enforcement of fisheries are costly and account for 25%–50% of the public expenditures on fisheries (Sutinen and Kuperan 1999). In some cases monitoring is financed by the fishing industry. For instance, in the U.S. halibut and sablefish fisheries, fishermen are required to enter catch information at landings on debit cards. Halibut fishermen are also required to record landings on fish tickets, which are checked against buyers' records. Capacity limitations and economic vulnerability of fishermen make this solution less feasible in developing countries.

Factors required for regulatory compliance include content of the regulation, its distributional impact, and its design and implementation. Absent any of these factors, the resource users might consider the regulation illegitimate (Jentoft 1989). Eggert and Lokina (2010) studied regulatory compliance in the

Tanzanian part of Lake Victoria fisheries, finding that the decision to comply is related to traditional deterrence variables like probability of detection and punishment (see Chapter 13), but also to legitimacy and social variables. In their study, a small but substantial part of the fishermen reacted neither to normative aspects nor to traditional deterrence variables, instead persistently violating the regulation and using bribes to avoid punishment. When regulation is perceived as illegitimate or the economic incentives to violate are large, violation can be ubiquitous. Since in many developing countries corruption is rampant (TI 2009), any attempt to increase enforcement might be offset by more frequent use of bribes. Enforcing officers may also accept bribes to avoid social disapproval, adding another level of complexity (Akpalu, Eggert, and Vondolia 2009).

The ability to accurately monitor catch levels and balance these against quota holdings is critical to the success of ITQs. Usually, regulators establish procedures to monitor and enforce the ITQs based on a system that tracks the flow of fish and fish products from the catcher to the purchaser and then reconciles these catches with quota holdings. Hatcher (2005) examines the effects of noncompliance on quota demand and the equilibrium quota price in an ITQ fishery. He finds that whereas lower quota prices are implied unambiguously by expected penalties (see Chapter 13), the expectation of penalties based upon relative violations of quota can produce higher quota prices than in a compliant quota market. Therefore, if both compliant and noncompliant firms are in the fishery, a shift in quota demand from compliant to noncompliant firms would result. The implications of these findings for ITQ fisheries management are potentially significant. If we do not observe falling quota prices, we cannot necessarily conclude that cheating in the fishery is not increasing. At the same time, increasing quota prices for a given TAC does not necessarily suggest increasing profitability in the fishery; they may, at least in part, be attributable to increased cheating. As a consequence, quota prices on their own may provide misleading signals: Caution should therefore be exercised when looking at quota prices to make fishery management decisions.

Conclusions

As the contours of the last wild frontier that the oceans represent begin to become apparent, issues of management, efficiency, and equity surface, and resource economists have important roles to play. The role played by economists in renewable resources policy over the past quarter-century, particularly for fisheries, may be both applauded and criticized (Wilén 2000). The insight that ecosystems are capital and must be analyzed as such is fundamental. Other major achievements include the incorporation of open access and property rights into models that helped make ITQs an important and successful instrument. However, other policy-relevant issues such as property rights,

monitoring, enforcement, and bycatches deserve more emphasis. Similarly, the creation of appropriate institutions, property rights, and policy instruments to address diverse ecological and social conditions with appropriate incentives for monitoring and reporting, fair distribution of benefits, and flexibility in the face of changing circumstances and complex ecology remains a great intellectual challenge to environmental and resource economists.

Successful ITQ systems illustrate the importance of property rights and restricted access. They can maintain sustainable stock levels as long as TACs are not set too high. ITQ programs typically result in less capital, more fish, higher quality and prices, and thus recovery of economic rents. Quotas allow catch to be reduced to optimal (or at least sustainable) levels. If the quotas are shares of a TAC, then they provide short-run flexibility to match stock variations. In the United States, the North Pacific Halibut Act of 1982 gave decisionmaking authority to Pacific Fishery Management Councils, which now set local annual limits for commercial catches.

ITQs foster responsibility and stewardship in the fishing industry, including increased research and cooperation with regulations, but disposal of bycatch is still a problem. New Zealand has added features to provide fishermen with incentives to not discard fish: the government will buy the bycatch at a set price, or fishermen may purchase quotas for the bycatch. Even with such options, the discarding of excess catch and low-priced fish appears to be a problem common to all ITQ systems (Copes 1986). It has not been a major problem in Iceland, but in tropical multispecies fisheries, it might be a considerable obstacle to ITQ implementation (Arnason 1996), as would the cost of monitoring.

Implementing an ITQ program risks concentrating wealth and creating concomitant social tension. Various allocation mechanisms can be designed to help meet social, cultural, and distributional objectives such as protecting regional fishing cultures. Some programs have sought to limit concentration by restricting trade. This approach also limits the efficiency gains, but distributional and regional concerns may make some restriction desirable. The U.S. National Research Council (NRC) recommends ITQs but says they are no panacea for all fisheries (NRC 1999). Instead, regional councils should decide whether to use them on a fishery-by-fishery basis. NRC recognizes the need for certainty but also cautions beneficiaries against viewing the ITQs as compensable rights; the government must retain ultimate stewardship of the resource and may, for instance, decide to limit fishing in certain areas to certain methods, and such restrictions should in no way lead to compensation. Finally, NRC emphasizes the need for fairness in addition to efficiency as well as requirements (such as "skipper-on-board") for limiting the accumulation of shares among too few parties.

Neither ITQ systems nor CPR management represents the optimal or definitive solution to all fishery management problems; fishery managers and communities have both instruments at their disposal, and a great deal of fine tuning is needed to adapt either instrument to specific socioeconomic and

ecological conditions. CPR management is likely to be important in developing countries with dramatic diversity and large coastal populations dependent on coastal fisheries, whereas ITQ systems are more applicable to large, highly developed industries in fisheries with less ecological diversity. In the future, more hybrid instruments may combine ITQs with taxes or fees, and more attention may be paid to the allocation and duration of quotas. In addition, a more general approach to management may focus on ecosystem and recreational values.

Supplemental Reading

Akpalu 2008

Colin et al. 2010

Hatcher and Gordon 2005

Hannesson 1998

Munro 2009

McGuire 1983

Nøstbakken 2008

Watson 1982

Notes

- 1 The impacts are large: Several tens of thousands of fishermen were made unemployed by the decline of cod fishing in Canada during the 1990s. However, most of the shortfall in income has been replaced by landings of shellfish (Arnason, Hannesson, and Schrank 2000).
- 2 There might be many intermediate examples where effective fishing methods entail some increased risk to fish populations, such as fine-meshed nets, which catch a high proportion of juveniles. However, some technology is simply effective, such as equipment to localize or lure fish to one location.
- 3 Taxing has been successful when the proceeds were used to “buy out” fishermen from the industry, thereby further reducing excess capacity and keeping the funds within the group of fishermen.

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Agriculture

Agricultural economics is a vast subject, and even the interface between agriculture and environmental issues is too large to cover in just one chapter. We have selected a few issues to highlight, primarily because of their importance and because they illustrate interesting applications of principles that are crucial for the design of environmental policy instruments.¹ Nonpoint-source pollution (NPSP; see Chapter 10) is a pervasive feature of agriculture.

The economics of poor households that depend on natural resources such as small-scale agriculture or forestry are typically very different from the simple neoclassical textbook case of a profit-maximizing firm. The case is not that they do not care to maximize profit, but missing markets and information, a lack of reliable titles, and constraints in various markets (labor, credit, insurance, and input and output markets) routinely lead to “corner solutions” in which the simple first-order conditions of maximization do not hold.

The inherent riskiness or stochasticity of all natural resources–based activities is an important factor in explaining the “nonneoclassical” feature of NPSP. Sharecropping (see Chapters 2 and 10) has been labeled “inefficient” because the tenant receives only a share of marginal benefit but has to bear the whole marginal cost (Marshall 1920). Thus, cost and benefit are not equated, as would be expected in simple standard optimization models. This result is explained by stochastic outcomes, together with disparities in income and risk adversity as well as difficulties with monitoring and the ensuing moral hazard.²

Other important factors are property rights, returns to scale, incentives, and efficiency. Many economists have believed that the poor farmers who typically are sharecroppers are too small to survive in agriculture (or forestry) and thus are doomed to become landless laborers or possibly commercial farmers (e.g., Sadoulet, Fukui, and de Janvry 1994). One consequence might be that the problems associated with sharecroppers are transitional; therefore policy should focus on large-scale farms (either through policies that favor private concentration or through state or cooperative farms, depending on one’s ideology). However, little definitive evidence supports this hypothesis (Binswanger and Holden 1998). Advantages to scale appear to be limited in most forms of agriculture (Berry and Cline 1979; Prosterman and Riedinger

1987; Rosenzweig and Binswanger 1994), and the incentive advantages of sharecropper agriculture may in some cases outweigh scale effects; however, excessively small, fragmented farms are inefficient. The prevalence of moral hazard gives an advantage to family labor over hired labor. Advantages to scale depend on geophysical and ecological conditions, among other things. For example, agricultural equipment might provide considerable advantages to scale in the plains but probably not in a hilly area.

Advantages to scale in the processing (as well as marketing and transport) of certain agricultural goods such as tea, sugar, palm oil, rubber, and coffee often have led to large-scale plantations of these crops. Close ties between harvesting and processing are natural but should not be inevitable. In fact, contractual arrangements between processors and small-scale farmers appear to be replacing the large plantations for sugar cane in Thailand and India and for oil palm in Indonesia (Binswanger, Deininger, and Feder 1995), and for tea in Sri Lanka (Herath and Weersink 2009). The processing industry itself is often extremely contaminating despite the fact that abatement may be technically easy.

Small-scale farming, which is so common in the world, may be the best way of managing agriculture and related resources. However, some problems such as farmers' experience may lead them to unsustainable behavior. One of these is tenure insecurity, which, together with other factors, may lead to lack of interest in soil conservation. Another such problem area is the nexus between high discount rates and credit rationing. Both issues are discussed later in this chapter.

There is an important distinction between irrigated and rain-fed agriculture. Irrigated agriculture is associated with fertilizers, pesticides, high-yielding varieties, and usually a high degree of mechanization. Water supply is a major issue (see Chapter 23), as are nutrient runoff, water quality, nutrient depletion, and the risk of salinization in some semi-arid areas. Reliance on a few high-yielding crops may decrease biodiversity and increase exposure to risk. Intensive pesticide use is associated with risks of damage to ecosystems and to the health of farm workers and their families. However, pesticide application may be a rational response to risk in an environment where institutions for insurance and savings are absent.³ Rain-fed agriculture, on the other hand, is associated with complex relationships among poverty, discount rates, hillside deforestation, soil erosion, and conservation as well as downstream effects, property rights, population pressure, and tenure.

The provision of infrastructure such as roads is a powerful instrument affecting land use. New roads may open up areas to agricultural exploitation (and the cutting of virgin forest). They generally facilitate the enforcement of property rights and thereby extend frontier access, pushing the managed system (e.g., farmland, grazing land, plantations) farther into the unmanaged area of natural forest. Roads also can have a large impact on the productivity of agriculture and other activities. They are generally high on the list of priorities of poor rural people. Increased accessibility to markets could conceivably more than compensate for any negative effects by increasing incentives

for organized and sustainable agriculture, silviculture, and land management in general. Extensive and intensive road networks must be differentiated. Whereas an extensive network of roads erodes incentives for sustainable forestry and agriculture (by providing new cheap land to be mined), an intensive network of farm-to-market roads makes small-scale agriculture viable (Schneider 1995). Agricultural incentives can have profound impacts on forests depending on the stage of development. In the first stage, they have a direct effect on the natural forest by encouraging the clearing of land. In later stages, they may lead to incentives for managed silviculture.

In the Amazon, the kinds of policy mentioned here, the classical policies (e.g., agricultural extension and the provision of related services), and the role of government structures interact in a complicated way (Schneider 1995). Four attractive policy recommendations can promote growth and resource conservation at the same time:

- eliminate perverse subsidies that artificially benefit deforestation,
- disseminate (e.g., through agricultural extension) technologies that increase the productivity of existing land,
- enable indigenous communities to manage their lands efficiently, and
- provide infrastructure (rural roads, small-scale irrigation, and so forth) that provides “safety nets” for poor communities (Deininger and Minten 2002).

The broader issues of land use allocation between agriculture and forestry are addressed in Chapter 27.

Labeling as a policy instrument has the potential to benefit or harm developing country producers in the future. It seems reasonable to presume that many producers in developing countries might benefit from ecologically sensitive production (partly because it is labor intensive). Several initiatives already exist, such as shade-grown coffee and organically grown fruit. However, the costs of certification may be too high for small-scale farmers in this context (Gobbi 2000). Harm might result from labels that are designed to suit northern consumers (or producers) and come to act as trade barriers.

Managing Agricultural Runoff

Agriculture releases significant amounts of several pollutants into the atmosphere. Regarding climate change, agriculture accounted for an estimated 10%–12% of total anthropogenic emissions of GHG. Of global anthropogenic emissions in 2005, agriculture accounted for about 60% of N_2O and about 50% of CH_4 . See Smith et al. (2007) and Box 26-1 for a discussion of the links between agriculture and climate change.

Agricultural runoff generally has a nonpoint-source character. Various externalities (e.g., soil, nutrients, pesticides, odor, genetically modified organisms,

BOX 26-1 Agriculture and Climate Change

The links between agriculture and climate change are twofold. On the one hand, agriculture is both a potential source of emission reductions and a potential sink which can offset the GHG emissions through carbon storage. On the other hand, changes in temperature and precipitation, as well as the increased frequency of extreme climate effects will affect agricultural productivity.

On the side of mitigation, agriculture's contribution to climate change varies between developing and developed countries. Developing country emissions largely arise from deforestation and land degradation, whereas developed country emissions are largely caused by fossil fuel-based emissions through energy use, reductions in soil carbon through intensive tillage, nitrous oxide emissions from fertilizers, livestock feeding, residue management, and tillage (McCarl and Schneider 2000). A variety of options exists for mitigation of GHG emissions in agriculture: reducing emissions by improved crop and grazing land management, restoration of soils that are drained for crop production, and restoration of degraded lands. These measures have clear synergies with sustainable development policies and could be implemented immediately using current technologies (Mayen et al. 2010).

On the side of adaptation, climate change will affect crops in several ways. First, changes in temperature and precipitation will alter the timing and length of growing seasons. Higher latitudes will benefit the most from longer growing seasons. In lower latitudes, though, higher temperatures are expected to adversely affect growing conditions—especially in areas where temperatures are at maximum levels for crop growth to begin with. Second, rising atmospheric CO₂ concentrations affect agriculture by increasing the rate of photosynthesis and water efficiency. The IPCC (2007) estimates that doubling CO₂ concentrations would lead to yield improvements ranging from 10% to 30%. Third, water availability is a critical factor in determining the impact of climate change. Although climate change prediction models suggest global mean precipitation increases of 3% to 15%, regional distribution is uncertain, and so are the related impacts on agriculture. Finally, agricultural losses can result from climatic variability and the increased frequency of extreme events such as droughts and floods; a higher frequency of droughts is likely to increase pressure on water supplies, while increases in rainfall intensity can lead to higher rates of soil erosion, leaching of agricultural chemicals, and runoff carrying livestock waste and nutrients into water bodies. Fischer et al. (2002) quantify the impact of climate change on agricultural GDP by 2080 as between -1.5% and +2.6%, with a large regional variation. Overall, mid to high latitudes will benefit, while agriculture in low latitudes will be adversely affected.

and pests) spread from one farm to another and to various downstream (or downwind) recipients, but the source (individual farmer or field) is typically difficult to pinpoint. Thus, designing policy instruments may be difficult; however, it need not be the case. For hazardous pesticides, for example, a tax, ban, or other regulation stands a good chance of being effective (unless there is smuggling).

For agricultural practices such as manure application, soil terracing, and the use of inputs that are not classified as hazardous, designing the proper instruments is more difficult but still important. The application of large quantities

of nutrients in intensive farming can be detrimental to the water bodies into which those nutrients ultimately flow. For instance, water bodies can become eutrophic—depleted of dissolved oxygen—a state that results in considerable loss in biodiversity and aesthetic as well as recreational and productive values.⁴ It is caused not only by synthetic fertilizers but also by natural manures, which cannot be taxed, and agricultural practices such as draining wetlands and straightening the watercourses of streams and rivers. These processes remove the natural opportunities for denitrification and other retention of nutrients. The restoration of wetlands can be more cost-efficient than the building of sewage treatment plants when it comes to nutrient removal to stop eutrophication (Gren 1991).

Many freshwater lakes and even some coastal and marine areas in Europe are severely eutrophic. In Africa, Lake Victoria—which drains a catchment area in which 30 million people live—is severely affected by problems ranging from overfishing to the invasion of foreign fish and plant species (Nile perch and water hyacinth, respectively). The lake receives enormous amounts of nutrients from various sources: soil erosion caused by deforestation and hill-side agriculture, agriculture close to the lake and on former wetlands that were drained, and poorly managed agricultural industries. Sugar plants, coffee mills, and other processing plants discharge large amounts of nutrients. The total surplus of nutrients in the lake is a vital factor behind the massive takeover of the water hyacinth, which in some years during the 1990s was so great as to stop not only fishing canoes but large ships and ferries. For long periods, the hyacinth mats were so dense that ships could not enter port and power stations had to be closed. The hyacinths also contributed to the spread of malaria and other diseases by providing habitat for mosquitoes as well as providing habitat for snakes and other dangerous animals.

The studies on NPSP discussed in Chapter 10 assume uncoordinated selfish behavior in a Cournot–Nash equilibrium. This assumption does not account for the possibility of interaction (nonmarket competition, rivalry, or cooperation) among farmers. Still, neighbors do collaborate and have information about each other, and this information needs to be brought into the analysis. Sociological and political science literature show that effective monitoring is often possible by peers even when it is not possible by outside authorities. This finding has led to the development of schemes in which farmers are encouraged to collaborate to protect the environment in a watershed. The tax rate may be lower in the case of full cooperation, and the polluters thus have an incentive to cooperate. (See Millock and Salanié 1997 for a model of an ambient tax with cooperation in abatement; and Shortle, Horan, and Abler 1998 for an overview of research issues in this field.) However, they also have an incentive to free ride, and thus the group must have peer monitoring as well as a credible punishment strategy that can sufficiently prevent the polluters from deviating from the cooperative group. The national environmental protection agency can choose a two-tier system in which farming

communities choose between tough conventional regulation (e.g., high taxes on pesticides) and this kind of collaboration. The result is to be monitored at a location such as a watershed, and if the collaboration fails, the environmental protection agency reverts to the conventional regulation as its “threat point.”⁵

If farmers can be assumed to observe each other's efforts to reduce pollution, then peer monitoring might minimize the information costs of regulation in applying an ambient tax to control agricultural NPSP (Byström and Bromley 1998). The incentive scheme, based on nonindividual contracts between farmers and the regulating authority, can achieve the cost-effective outcome even if farmers do not have homogeneous risk preferences (i.e., there is adverse selection). Also, because farmers are allowed to trade pollution abatement efforts, neither individual contracts with each farmer nor full information on the part of the regulatory agency are necessary to achieve cost-effective abatement, which will substantially lower the information costs of regulation. This result is more efficient than the (conditional) subsidies scheme to overpay the agents due to imperfect agricultural monitoring. The only weakness is that the system of collective penalties might have weak legal support because criminal law in general requires that liability be established at an individual level.

Common property resource (CPR) management is another relevant mechanism in this context because it has the potential to overcome the incentives for free riding and the costs of monitoring by using the cohesion of culture together with specially designed rules of management in a way that can be (at least partly) modeled in terms of repeated games. Peer monitoring is often superior to monitoring by agents of an outside authority (Ostrom 1990). Because peers have more relevant information and more legitimacy than outside monitors, monitoring is cheaper, more accurate, and perceived as being more fair. CPR management appears to be particularly useful for the marginal resources in relatively poor contexts in which yields are so low as to make the cost of fencing appear high or so variable as to make them inappropriate. (See additional examples and discussion in Chapters 23 and 28.)

One of the critical environmental issues in agricultural policy is how to compensate farms that provide additional ecosystem services (Lankoski et al. 2010). In many industrialized countries, the agricultural sector is subsidized, and farms may receive subsidies that are explicitly tied to agricultural practices that reduce runoff (e.g., leaving strips along waterways, planting certain crops, keeping herd sizes relative to pasture, letting land lie fallow, and timing and zoning fertilizer application). In poor countries, agricultural subsidies are typically less elaborate, and in general, the agricultural sector is taxed heavily (not necessarily directly but through the setting of relative prices). Compensation for ecosystem services—fairly new in developing countries—can in principle be funded by a share of ecotourism; hunting or fishing revenues; contracts with bioprospecting companies; increased revenue from eco-labeled products; or direct payments from parks, public authorities, or other downstream users (e.g., Pagiola 2000; Ferraro and Kiss 2002). Many

sources of compensation for ecosystem services are likely to target forest practices—for example, Costa Rica has one of the most elaborate systems of direct payment for reforestation, forest conservation, and sustainable forest management activities (see Chapter 28)—but some agricultural activities also may be included. For instance, increasing concerns over the environmental impact of agriculture in the European Union led to the introduction of agri-environment schemes starting in 1992. These schemes compensate farmers financially for any loss of income associated with measures that aim to benefit the environment or biodiversity. Main objectives include reducing nutrient and pesticide emissions, protecting biodiversity, restoring landscapes, and preventing rural depopulation. In virtually all countries the uptake of schemes is highest in areas of extensive agriculture where biodiversity is still relatively high and lowest in intensively farmed areas where biodiversity is low. Unfortunately, few studies assess the effects of agri-environment schemes on biodiversity conservation. The limited number of studies that have been published present contrasting results (see Box 26-2).

BOX 26-2 Payments for Ecosystem Services in Agriculture in the European Union

One response to concerns over biodiversity loss has been the introduction of agri-environment schemes, in which farmers are paid to modify their farming practice to provide environmental benefits (Swinton et al. 2007; Klemick 2011). The EU explicitly addressed the impact of agriculture on the environment in 1992, when EEC Regulation 2078/92 was introduced, requiring all EU member states to apply agri-environment measures in environmentally sensitive areas (ESAs) according to environmental needs and potential.

Agri-environment programs vary markedly among European countries. Their objectives usually reflect a combination of the agriculture's environmental, ecological, and socioeconomic problems and—perhaps more important—the politics and vested interests of each country. In Switzerland, the Netherlands, and the United Kingdom, schemes available to farmers concentrate on wildlife and habitat conservation. In Denmark and Germany most schemes offered to farmers aim to reduce agrochemical emissions, while the French program is geared toward preventing land abandonment in agriculturally marginal areas. In Ireland and Austria, program objectives balance environmental protection, biodiversity conservation, and landscape maintenance.

Schemes can be implemented either horizontally throughout the country or zonally in certain areas identified as particularly vulnerable or local biodiversity hotspots. Governmental bodies usually designate the areas where zonal measures can be implemented. Most countries combine both country-wide and zonal approaches, because a limited set of zonal schemes exist that aim to conserve vulnerable ecosystems (Kleijn and Sutherland 2003).

A few studies of these programs show positive effects measured in increased species diversity or abundance. However, other studies show no effects, negative effects, or positive effects on some species and negative effects on others; a number of schemes have not achieved the expected effect on conservation or even have negative side-effects. The main explanation seems to be the lack of correlation between payment criteria and biodiversity conservation (Kleijn and Sutherland 2003).

Land Markets, Soil Erosion, and Population Growth

Land is a crucial resource in agrarian economics,⁶ and efficient land markets (sales and rentals) are important for sustainable management and agricultural development (Benin and Pender 2009). Economic theory postulates three links between tenure security and agricultural economics incentives (Besley 1995). The first link, referred to as the “security argument,” captures the direct and positive link between tenure security and investment incentives. The second, called the “collateral-based argument,” is based on the premise that when land tenure is secure and thus easier to collateralize, it can reduce the price of capital and subsequently increase the value of investments. Finally, the “gains-from-trade argument” notes that tenure land rights expand trading opportunities and the ability to take advantage of gains from trade by lowering transaction costs if land is to be either rented out or sold.

Unfortunately, land sales markets are usually poorly developed in developing countries due to asymmetric information about land quality, lack of land titles, undeveloped credit markets, inability of poorer farmers to pay the collateral value of the land, and various policy distortions. Where land sales markets do not function well, or are prohibited—as in Ethiopia, Eritrea or Zambia—the effective functioning of land rental markets is very important for agricultural development. Land rental markets serve an important role as a safety net for poor landlords in many places in Africa; food and income from rented out land are very important, particularly for poor female-headed households that lack the capacity to farm the land themselves. In a sense, they promote short-term agricultural efficiency by allowing land to be used by farmers who are more capable of earning the highest return from it; however, if rental contracts are very short-term due to tenure insecurity of landlords, the incentives of tenants to invest in sustainable land management practices may be very limited.

A large literature has investigated the short-term efficiency impacts of land lease contracts (see for example, Holden et al. 2009). Benin and Pender (2009) investigate the links between the short-term nature of rental contracts and sustainable land management in the context of northern Ethiopian highlands. Compared to other countries, land is relatively equally distributed in the Ethiopian highlands because of a radical reform program begun in 1975. Nevertheless, significant differences exist across regions when it comes to tenure security. For example, one village included in this study has experienced fourteen land reforms since 1975. The results of this study highlight how land tenancy affects the local context and policies, suggesting that the rental market functions better in regions where tenants feel sufficiently confident about their long-term ability to renew their lease contracts. In such settings, they have incentives to make investments in land improvements and to adopt more sustainable land management practices, despite the short-term nature of the contracts. In contrast, in regions where landowners fear further

redistributions, tenants appear to have a shorter-term perspective, and the ability to assure sustainable land management seems more compromised.

Zikhali (2010) found some similar results in the case of Zimbabwe. Since its independence in 1980, Zimbabwe has pursued a land reform and resettlement program aimed at addressing racially skewed land distribution. The most recent phase, the Fast Track Land Reform Program (FTLRP) launched in 2000, had a goal of extensive and compulsory land distribution. In principle, the land tenure system under the FTLRP offers each household a long-term lease and the option to purchase the land, but in practice there are many different types of temporary licenses which the government intends to convert to permanent leases in the future. Zikhali (2010) investigates the impact of this program on perceptions of tenure security, finding that the program not only created some insecurity among its beneficiaries, but it also had an adverse impact on investments in soil conservation. Investment in land conservation depends on other factors as well (Dasgupta and Mäler 1995; Lutz, Pagiola, and Reiche 1994; Shiferaw and Holden 1999). One of the most contentious and serious issues is the effect of population pressure. The pessimist Malthusian view is that increasing population will lead to falling marginal productivity, overuse, and resource degradation, further undermining human health and deepening poverty (Cleaver and Schreiber 1994). Policy responses reflecting this view emphasize population control, environmental education, and controls on access to resources. The optimistic Boserupian view describes how increasing population pressure is the mechanism that leads to improved technology and increased efficiency in land management and thereby prosperity (Boserup 1965). Evidence can be found to support both positions (Binswanger and Deininger 1997; see also Heath and Binswanger 1996; Sterner and Segnestam 2001; and Tiffen, Mortimore, and Gichuki 1994).⁷ A review of more than 70 empirical studies in poor hill and mountain regions was inconclusive about the effects of population growth on land and forest quality (Templeton and Scherr 1999). Other underlying factors must explain why the results are so dramatically different.

A study of Machakos district, Kenya, is a prime example often quoted in support of the Boserupian hypothesis (Tiffen, Mortimore, and Gichuki 1994). Despite a 500% increase in population over 60 years, the land degradation and food insecurity that was predicted for this area in the 1930s has not occurred. Instead, food production has increased manifold due to a threefold increase of livestock holdings per farm and a doubling of the total cropland area. Considerable local investments in water and soil conservation provide the conditions for horticulture and small-scale dairy production. The farmers organize themselves in efficient democratic workgroups (*mweya*) and enjoy secure land rights, infrastructure, and market access on favorable terms—partly because of the proximity of Machakos to Nairobi.

In other areas of Kenya and in the Ethiopian highlands, several of these vital factors are missing; as a result, high population pressure appears to be

positively correlated with soil degradation (and increased poverty). In a study of Muranga district, Kenya, between 1960 and 1996, population increase is associated with more soil erosion, not conservation (Ovuka 2000).

Soil conservation practices and increased agricultural production are more easily achieved in communities that collaborate democratically than in those where each farmer works individually. Many studies show the importance of policies, property rights, and good democratic institutions in avoiding the Malthusian spiral of increased population, poverty, and resource degradation (e.g., Ekbom, Knutsson, and Ovuka 2001). Land conflicts may also harm soil management. A recent example is the post-presidential election violence in Kenya in December 2007, where historical land conflicts between ethnic groups amplified the post-election violence. In Kenya, land rights have been in place since independence. However, the high cost of registration has discouraged updating the registrations after land transactions such as inheritance and sales. Yamano and Deininger (2009) found that pending conflicts and concerns about future conflicts reduce the use of inputs in agriculture in Kenya and that land conflicts should be solved to sustain land quality. On conflict resolution, they find that people resort to informal institutions first and then to formal institutions only when the informal institutions are unable to resolve the conflicts; moreover, fewer pending conflicts occur in communities with more elder groups. Thereby, in order to improve tenure security in a sustainable manner, land titles must not only be provided but also updated. In addition, the roles of local informal institutions in protecting land rights must be recognized.

Risk in Sharecropper Agriculture

Whereas tenancy exists all over the world, its arrangements take various forms. In the fixed-rent variety, the tenant pays a fixed sum of money to the landlord in return for the right to cultivate the land. In contrast, with sharecropping, the tenant yields to the landlord an agreed-upon share of the crop. Sharecropping is an arrangement that has particular value when the tenant is small and averse to risk: If a given fraction of output is paid as rent, then the tenant is to some extent insulated against output fluctuations because he can share some of these fluctuations with the landlord (Ray 1998).

A long tradition in economics argues that sharecropping is not as efficient as the fixed-rent tenancy scheme. Sharecropping leads to undersupply of the tenant's inputs since the tenant does not keep the entire marginal output resulting from his effort (see Chapter 2). However, this inefficiency relies on the assumption that the use of inputs by the tenant—such as labor—cannot be perfectly monitored and enforced by the landlord. If perfect monitoring were possible, the form of the tenancy contract would be irrelevant from an efficiency point of view.

Sharecropping enjoys popularity in real-world practice despite any theoretical inefficiencies. Surely, there is more to the story than currently has been

outlined. For example, if the landlord is deeply involved with the crop grown of the land, the methods used for cultivation, and the maintenance and care of the plot, sharecropping leaves the landlord in the position to make suggestions, to provide managerial care, and supply inputs of production. Some of these inputs may be noncontractible, just as the tenant's labor is noncontractible because it cannot be observed or verified by the landlord. Sharecropping may be a compromise solution in which both the landlord and the tenant put in some effort.

Kinship might increase the inefficiencies of sharecropping. For example, in a study about the relationship between kinship and land productivity, Kassie and Holden (2009) assess the output levels and fertilizer input levels of kin and nonkin sharecropped tenants' plots in Ethiopia. As in many other African countries, restrictions on land renting were practiced in Ethiopia for many years. Radical land reform in 1975 prohibited land renting and hiring of labor. Although the current regime is more market-friendly, all land remains state-owned, land sales are prohibited, only short-term renting is allowed, and sharecropping is very popular. Kassie and Holden (2009) differentiate sharecropping into kin and nonkin sharecropping tenants and control for self-selection due to unobservable household characteristics and a range of plot-level characteristics. They find significant signs of inefficiency. Efficiency on rented land was higher than that on owner-operated land of the same tenant households; tenants used more fertilizer on their rented plots than on their own, as this would enhance their probability of rental renewal. Thus, the results indicate that the threat of eviction offsets the inefficiencies of sharecropping to a large extent. However, kinship reduces the threat of eviction since it is more difficult to evict kin tenants than nonkin tenants, implying that nonkin tenants feel less secure about their tenure and therefore use more inputs on sharecropped plots to increase the probability of renewal. Holden and Bezabih (2009) found a significantly higher level of inefficiency linked to contracts of female landlords with in-law tenants, which may be due to the high eviction costs of tenure-insecure female landlords who typically live in the village of their in-laws and are therefore less able to freely screen and select better tenants.

Asymmetries of risk and resources can lead to many ecologically unsustainable and apparently counterintuitive behaviors. One famous example of course is holding herds that are too large (because no other means of banking is available), causing overgrazing that leads to serious pasture shortages and massive cattle death in times of drought. The following section gives details of another example involving excessive use of pesticides.

Crop Insurance as a Substitute for Pesticides

"As the tide of chemicals born of the Industrial Age has arisen to engulf our environment, a drastic change has come about in the nature of the most serious public health problems. Only yesterday mankind lived in

the fear of the scourges of smallpox, cholera, and plague that once swept nations before them. Now, our major concern is no longer with the disease organisms that once were omnipresent: sanitation, better living conditions, and new drugs have given us a high degree of control over infectious disease. Today we are concerned with a different kind of hazard that lurks in our environment—a hazard we ourselves have introduced into our world as our modern way of life has evolved.” (Carson 1962, 168).

In this quotation, Rachel Carson alludes to the danger of uncontrolled pesticide use. She goes on to give example upon example of humans and birds whose health, reproductive capabilities, and even lives have been destroyed as a by-product of the use of dieldrin and other chemicals, many of which have since been banned, some almost directly as a result of the publication of her book, *Silent Spring*. Still, many thousands of people are poisoned each year by pesticides and other hazardous chemicals. In Brazil alone, more than 5,000 cases of pesticide poisoning were reported in 1998, and according to the World Health Organization (WHO), for every case of poisoning reported, an estimated 49 are not (Lins 1996).

One striking example of how pesticides are used where other methods might be preferable is in the fight against desert locusts. The U.N. Food and Agriculture Organization (FAO) spends millions of U.S. dollars yearly on spraying pesticides to minimize the threat of locust infestations in Africa. Yet the actual average damage to crops is small. In some 50 years of data, a relationship between harvest loss and locusts is not apparent (in fact, the relationship is sometimes inverse because the locusts tend to “attack” in years of good harvest). Expected harvest loss is minimal and, according to some authors, lower than the cost of pesticide spraying (Heroc and Krall 1995; Joffe 1995). In addition, the widespread use of pesticides has been linked to ecosystem effects and other costs.

However, because infestations tend to be locally concentrated, an individual farmer’s crops could be almost totally destroyed by desert locusts. In the absence of insurance, income diversification, and banking services, such risks are ultimately fatal and thus unacceptable. Hence the urgent need for relevant, effective policy instruments and the reason why governments support the FAO pesticide strategy. However, the convention of pesticide use against desert locusts is being rethought, partly because the locusts are considered a delicacy in the Sahara region but lose their sale value if there is a risk that they have been sprayed. Presumably, farmers in the affected regions would be better served by insurance than by pesticides (Smith and Goodwin 1996). The question is whether insurance companies would be willing to insure farmers given the moral hazard inevitably involved in crop insurance. If donors or governments can help set up the institutions that facilitate the provision of insurance (maybe insurance against desert locust infestations, which would be easier to verify than just crop failure in general), then the need for pesticides would be drastically reduced.

Genetically Modified Crops (GMCs)

The term GMC refers to crop plants that have been modified in the laboratory to enhance desired traits such as increased resistance to pests, diseases, herbicides or improved nutritional content. Currently, GMCs are being widely used. For instance, in 2008, GMCs were being grown on 9% of the global arable land (James 2008). At present, only a few first-generation technologies to enhance herbicide tolerance and insect resistance in soybeans, maize, cotton, alfalfa, canola, and sugar beet have been commercialized. Second-generation technologies in the pipeline include product quality improvements for nutrition and industrial purposes while third-generation crops involve molecular farming where the crop is used to produce either pharmaceuticals, vaccines, or industrial products (such as enzymes and biodegradable plastics). The U.S., Argentina, Brazil, Canada and China account for most of all transgenic crops grown. Even though GMCs have been most rapidly deployed in industrialized countries, in recent years transgenic crop areas in developing countries have increased at a rate about four times higher than in the industrialized countries (Thies and Devare 2007).

GMCs are claimed to have the potential to solve many of the world's hunger and malnutrition problems (Bouis 2007), and to help protect and preserve the environment by increasing yield and reducing reliance upon chemical pesticides and herbicides. Nevertheless, consumers and policy-makers in many countries have reacted strongly against them due to concerns about potential health and environmental risks. Use, cultivation, and sale in the European Union are severely restricted and in some applications or countries banned altogether. The two main health risks under discussion are tendencies to provoke allergic reaction (allergenicity) and gene transfer. Despite the possibility that introducing a gene into a plant might create a new allergen or cause an allergic reaction in susceptible individuals, no allergic effects have been found relative to GM foods currently on the market (WHO). On the other hand, gene transfer from GM foods to cells of the body or to bacteria in the gastrointestinal tract would cause concern if the transferred genetic material adversely affects human health. This would be particularly relevant if antibiotic resistance genes (third-generation technologies), used in creating GMOs, were to be transferred. Although the probability of transfer is presumed to be low, the use of technology without antibiotic resistance genes has been encouraged by a recent FAO/WHO expert panel.

When it comes to environmental risks, the mixing of crops derived from conventional seeds with those grown using GM crops—i.e. “outcrossing”—may have an indirect effect on food safety and food security. This risk is real, as was shown in the StarLink incident when traces of a maize type which was only approved for animal feed use appeared in maize products for human consumption in the U.S. To avoid this unintended effect, several countries have adopted strategies to reduce mixing, including a clear separation of the

fields within which GM and conventional crops are grown. Other risks are the persistence of the gene after the GMC has been harvested; the susceptibility of nontarget organisms (e.g., insects which are not pests) to the gene product; the stability of the gene; and the reduction in the spectrum of nongenetically modified plants, involving a loss of biodiversity (WHO 2010).

Although the GM products currently on the international market have all passed some risk assessments, these have been called into question, and different GM organisms include different genes inserted in different ways. This means that individual GM foods and their safety should be assessed on a case-by-case basis and that it is not possible to make general statements on the safety of all GM foods. Governments around the world are working hard to establish regulatory processes to monitor the effects of and approve new varieties of GM plants. Different governments are responding in different ways, depending on the national political, social, and economic attitudes towards GMCs. Public reservation, particularly strong in Europe, has spread to other countries through trade regulations, public media, and outreach efforts of antibiotech lobbying groups (Qaim 2009). However, consumer acceptance of second-generation products seems to be larger due to the direct consumer benefits (Lusk et al. 2005). Some evidence also suggests that consumers in developing countries see GM food more positively than their counterparts in developed countries (Kimenju and De Groote 2008).

Countries which have legislation in place focus primarily on assessment of risks for consumer health and labeling. For instance, Europe requires mandatory food labeling of GM foods in stores. Other countries such as the United States have adopted a voluntary labeling strategy. In this regard, production and trade-related interests seem to play a major role in decisionmaking about labels. According to a recent study by Gruère et al. (2009), countries producing GMCs are more likely to have less stringent labeling policies. In contrast, food and feed exporters to the European Union and Japan are more likely to adopt labeling policies.

Several studies have analyzed the farm-level impacts of the introduction of GMCs (see Qaim 2009 for a survey). In terms of the yields, no significant difference between herbicide-tolerant and conventional crops has been found. One possible explanation is that, although GMCs reduce the cost of production through lower expenditures for herbicides, labor machinery, and fuel, the fee on seeds, which varies across countries and crops, seems to be of a similar magnitude or sometimes higher than the average cost reduction; thus, the gross marginal effect is very small or negative. However, the average farm-level profits seem to increase over time, partly as a result of seed-price adjustments and farmer-learning effects (Fernandez-Cornejo et al. 2005). On the other hand, insect-resistant crops have been shown to have significant income-increasing and poverty reduction effects. Farmers in developing countries seem to benefit more than farmers in developed countries due to weaker intellectual property regulations, and thus lower seed prices (Qaim 2009).

If their safety issues and potential negative effects can be resolved, GMCs might contribute significantly to food security and sustainable development at the global level. Nevertheless, if only rich farmers were to have access to GMCs, inequality would increase. Thus, institutional settings that provide information and credit and reduce infrastructure constraints are needed as well.

Financial Institutions To Reduce Risk

To some degree, insurance and banking are interchangeable options. Savings may act as a buffer that reduces the need for insurance. Even diversification of income or the portfolio of assets results in less variance and thus reduces risk and may act as a substitute for savings or insurance. All the factors that affect risk must be considered in one context. Village money lenders are known to charge exorbitant interest rates, which effectively amount to as much as 100% or even several hundred percent per year (for India, see Pender 1996; for Ethiopia, see Holden, Shiferaw, and Wik 1998, who find that many farmers live with rates of *more* than 100%). Such rates are typically the result of severe rationing in the credit market in conjunction with absolute poverty. The practice is detrimental to progress because many good projects will not get funded at this level of interest.

Why do other banks and bankers not enter the market and thus bring interest down to more reasonable rates? Why do international banks not move in, for instance? The primary reasons are high risk, economies of scale, asymmetric information, asymmetric monitoring, and high costs of information. Local money lenders incur low administrative costs commensurate with the small scale of lending, have better information about which individuals are bad credit risks, and can monitor their professional effort and performance. Thus, a local money lender can address all the major asymmetries—the moral hazard and the adverse selection problems. Together with freedom from economies of scale and perhaps other barriers to entry (e.g., various indivisibilities, startup costs, and the cost of communication), the local money lender essentially has a monopoly on the lending market, which reinforces the high interest rates and the credit rationing that is so detrimental to development and recovery from bad harvests and deaths as well as the costs associated with weddings, childbearing, illness, and education.

Several cooperative bank movements have been created to overcome such problems. The Grameen Bank in Bangladesh is one of the more successful (see Box 26-3).⁸ In the Grameen Bank, entrepreneurs must form groups with separate projects (which helps spread the risks) and apply for loans as a group (which reduces management costs). When loans are approved, typically only the first two members receive loans. If they manage to meet their repayment schedules, then two more members get loans, and so on until the last member, who is typically the group leader. The structure of lending around groups creates incentives for cooperation within the group and the village as a whole, and it improves monitoring of entrepreneurial activities and repayment

BOX 26-3 Grameen Bank

In 1976, Professor Muhammad Yunus of the University of Chittagong, Bangladesh, launched an action research project on the design of a credit delivery system to provide banking services to the rural poor. In October 1983, the Grameen Bank Project became an independent bank. The bank is mainly owned by the borrowers (90% of shares), with the remaining 10% owned by the government.

Grameen Bank (*gram* means “rural” or “village” in Bengali) has been very successful at providing banking services to clients who normally are thought of as the most difficult: the rural poor. According to its balance sheets, Grameen Bank’s total assets have grown from US\$5 million in 1983 to US\$1.5 billion in 2005. The bank has provided loans to about 7.5 million people in 80,000 villages, and has financed more than half a million houses. During the 1990s, the bank was converted into a “movement” with separate funds and organizations that promote irrigation, health, fish farms, and health and education programs. The bank actively seeks out the most deprived in Bangladeshi society—beggars, illiterates, widows—and still succeeds in attaining a loan repayment rate of 99%. It does not look or operate like other banks, and works through village meetings with the borrowers—mainly women, because they are considered more responsible than men.

The philosophy of the Grameen Bank is as follows:

Over the years, representatives of the borrowers have agreed with the bank on certain principles and commitments which they will undertake to help improve their lives and their ability to meet their debts. To Westerners these may seem at best paternalistic; however, the slogans are chanted enthusiastically by the borrowers. They pledge to abide by “the 16 decisions,” a set of personal commitments such as “We pledge to send our children to school,” and “We pledge not to demand or pay dowry for our daughters’ marriage.” The most important of these commitments is to join up with four fellow borrowers, none of whom can be a family member, to form a “group.” The group dynamic provides a borrower with the self-discipline and courage needed to enter into these uncharted waters. Peer pressure and peer support effectively replace collateral: if one borrower defaults the whole group is penalized. The system also saves the bank the costly business of screening and monitoring borrowers.

In 2006, Yunus and the Grameen Bank jointly received the Nobel Peace Prize for their achievement in assisting the rural poor.

Source: Grameen Communications (1998) and www.grameen-info.org (2011).

behavior reciprocally within the group. This approach also gives group leaders a clear incentive to make the system work (somewhat akin to the practice of giving leaders of irrigation cooperatives a stake in the fields that are last in the line of downstream plots).

Starting banks and insurance companies is not easy. It requires time, patience, and local knowledge and thus is perhaps a task that is best left to the local level. National policymakers (or donors) can facilitate this kind of

development through training and education and by creating the necessary institutional preconditions—at least removing the barriers that in many cases make the creation of such institutions difficult. Secure land tenure is essential in this context because it reduces risk directly, creates incentives for productive investments, and can encourage and facilitate banking (because land is a valuable form of collateral). Because poverty and insecurity may lead to many unsustainable and short-sighted practices, the provision of banking and insurance may be important for sustainable development.

Eco-Taxes in Agroindustry

A common source of pollution in many developing countries is the organic waste from sugar plants, paper mills, palm oil mills, and waste-producing plants that process coffee, tea, and other crops.⁹ The waste is biological in origin and, under appropriate conditions, biodegradable. It is not in itself hazardous; however, dumped in large quantities in an inappropriate location (such as a small river or water body), it can effectively kill the existing ecosystem through eutrophication, because its decomposition uses up all the oxygen available in the ecosystem. A measure of oxygen demand such as biological oxygen demand (BOD) is appropriate as a simple measure of this kind of pollution.

Malaysia recovered from the collapse of the rubber market partly by diversifying rather massively into oil palm plantations in the 1960s. The effluent wastes (roughly 2.5 metric tons per ton of oil) were pumped into the nearest water bodies. By 1977, “42 rivers in Malaysia were so severely polluted that freshwater fish could no longer survive in them” (Aiken and Leigh 1992). Not only did the fish die; freshwater supplies for households, villages, and even municipalities became unusable. Wells had to be dug. Oxygen levels were so low as to encourage anaerobic decomposition, producing hydrogen sulfide, ammonia, and other toxic compounds. The stench was reportedly so bad that entire villages had to relocate, and the mills became a major object of citizens’ complaints.

The Malaysian Department of Environment (DOE) could not envision stopping the expansion of the sector that already accounted for around 15% of export earnings in 1975 and was still in rapid expansion, providing both foreign exchange and employment opportunities. However, neither could the agency ignore the enormous effluent problem that this profitable industry presumably should have been able to tackle. The pollution load from the mills in 1975 was equivalent to the raw sewage from 12 million people (the population of the whole country).

The government used a combination of policies to tackle this issue and similar problems in the rubber industry. In 1974, the Environmental Quality Act was passed, which authorized the Malaysian DOE to require firms to be licensed and empowered the agency to set appropriate preconditions for licensing. The Malaysian DOE had the power not only to mandate technology and emissions levels but also to close plants and to differentiate license

fees, making it in effect possible to levy fees that were akin to environmental charges. The agency adopted a consensus-building approach by forming an expert committee involving representatives from the industries concerned. This committee and the individual industries searched for possible treatment technologies, and within two years, the agency had seen sufficient progress to feel confident in formulating its conditions.

In July 1978, the Malaysian DOE announced its palm oil regulations. Three policy instruments were used:

- a progressive system of standards that applied to the effluents,
- a varied license fee that corresponded closely to a two-tier effluent charge, and
- subsidies for abatement technology and research into such technology.

The basis for the system was the standards, which applied to several effluent parameters, the most important of which was BOD. The required improvements in effluent were dramatic—reductions of several orders of magnitude resulted within less than a decade (see Table 26-1). The agency did not make the achievement of these standards mandatory the first year in order to give the plants time to make the necessary abatement investments. To stimulate those investments, the agency made the license fee variable, with a small fixed component and a variable component that had two tariff levels, one below the standard and a higher charge above the standard:

$$T = T_0 + T_1\hat{e} + T_2(e - \hat{e}) \quad (26-1)$$

where e is actual effluent; $\hat{e}$ is the allowable effluent according to the standard; T_0 is the fixed charge; T_1 is the standard fee for allowable pollution; and T_2 is the “penalty” charge for “excess” pollution, set at ten times the charge for allowable pollution (T_1).

TABLE 26-1 BOD from Effluents and Required Standards for Palm Oil Mills

<i>Standard</i>	<i>Year</i>	<i>BOD (ppm)</i>
None	Before 1977	25,000
A	1978	5,000
B	1979	2,000
C	1980	1,000
D	1981	500
E	1982	250
F	1984	100 ^a

Notes: BOD = biological oxygen demand in parts per million (ppm); None = untreated effluent, typical value. ^aEven 20 ppm under some conditions.

After the first year, the standards became binding, and the Malaysian DOE demonstrated its statutory powers by closing one plant in late 1979 and a few more between 1981 and 1984. Presumably, this action reduced the role of the fee, although the fee still had to be paid, and the agency used its discretion to allow some firms to continue operation despite violating the standards as long as the fees were paid. This discretion was applied to plants that were perceived as making serious efforts in the right direction, and plant closure (refusal to renew licenses) was used as a threat for the more recalcitrant firms.

The third instrument that the Malaysian DOE used was a form of subsidies for research and development (R&D) in abatement technologies. Actual funds were not paid, but under special circumstances (and with the approval of the minister of the environment) the above-mentioned fees could be reduced or waived as a quid pro quo for R&D investments of general interest. The government also established the Palm Oil Research Institute of Malaysia, which increased the capacity to find new abatement technologies.

The total results of all these policies are presented in Figure 26-1, which illustrates how effluents were reduced rapidly despite a continued expansion of the palm oil industry (as shown by the upper curve for total BOD produced). Disaggregating the effects into separate components for the fee, the standards, and the technology subsidies is tricky. In the first year, the fee may have been an important element of the policy, but because plants faced the risk of closure, physical standards presumably became more important. The fees would still act as an additional incentive and as a way of leveling the playing field between the more compliant (with higher abatement costs) and the (somewhat) less compliant plants. A survey carried out in 1980 and 1981 showed that the effluents of 90% of the mills were below Standard D—500

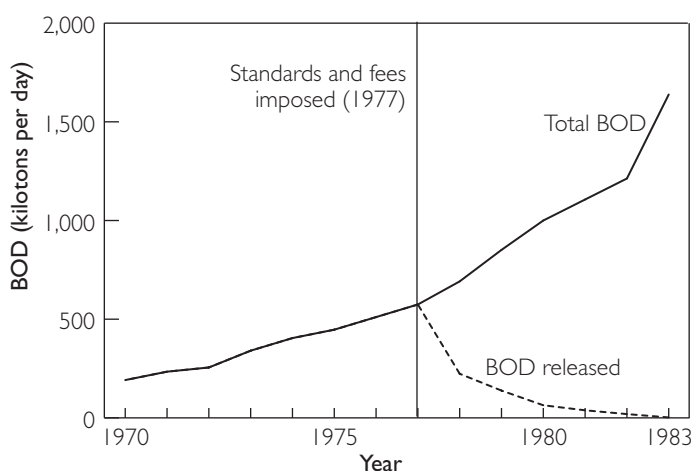


FIGURE 26-1 Production and Release of BOD Effluent from Malaysian Palm Oil Mills

Source: Adapted from Vincent and Ali 2005.

parts per million (ppm)—which was applicable starting in July 1981, and the effluents of 40% of the mills were below 100 ppm—a standard that would not come into force until several years later. These extreme reductions could be interpreted as a sign that the fee was still having an incentive effect, although it could just as well be due to the indivisibilities in technology, suggesting that firms prefer to aim for future targets to avoid more costly investments later.¹⁰

The success of the policy package has been attributed to its combination of policies. A pure effluent fee probably would not have been sufficient because several plants would have preferred to pay the fee, creating unacceptable local pollution conditions in some areas (Vincent and Ali 2005). The timing of the standards and tax subsidies for R&D was the result of industry influence (exercised through the consultative process in the expert panel set up by the Malaysian DOE) and was crucial in securing industry acceptance and support.

Finally, technological progress was an important factor. The nontoxic and biological nature made the problem relatively easy to solve. Several cheap and, in some cases, even profitable abatement technologies have been discovered (Mayen et al. 2010). The biomass and sludge are excellent raw material for animal feed, fertilizer, biogas, and several other products that have been commercialized successfully. The Malaysian experience should encourage those grappling with similar problems, such as bagasse and other biological wastes from raw material processing—and even the water hyacinths of Lake Victoria.

Supplemental Reading

Byerlee et al. 2009	Ray 1998
Hanley and Oglethorpe 1999	Rosenzweig 1988a, 1988b
Mendelsohn and Dinar 2009	Sexton et al. 2007
Pingali 2007	Swinton 2010
Randall and Taylor 2000	

Notes

- 1 Relevant topics not covered due to space constraints include: agricultural water, which is often grossly subsidized and badly mismanaged (see Chapter 23); issues involving subtle externalities (see Brown and Layton 1996 on antibiotics resistance as a depletable resource); and public goods together with difficult ethical and political questions. Trade barriers and trade liberalization are typically vital issues for agriculture in developing countries (see Chapter 14 for a discussion of the underlying principles and Anderson 1998 for an overview).
- 2 This is one example of the interlinked markets—a linkage among labor, land, and insurance. Other examples are found where markets for credit, inputs, output, savings, insurance, land, and labor are linked in various ways (surveyed in Otsuka, Chuma, and Hayami 1992).

- 3 For further analysis, see Zilberman and Millock 1997; Zilberman, Templeton, and Khanna 1999; or Wu 1999 on crop insurance and nonpoint-source pollution. See Wu and Babcock 2001 for a focus on spatial heterogeneity and instrument choice.
- 4 Farming practices may have many other downstream consequences: risk of landslides and flooding, the desiccation of rivers or lakes (e.g., the Aral Sea, emptied largely by massive irrigated cotton farming), salinification, desertification, and the destruction of river ecosystems and even coral reefs.
- 5 Such a combination has been proposed to improve water quality, with a background threat of mandatory controls or taxes if the voluntary approach is unsuccessful in meeting a previously specified water quality goal (Segerson and Miceli 1999). The regulator uses a uniform subsidy rate (i.e., one that does not depend on farm characteristics) to induce participation in a voluntary program to achieve a given water quality goal. This suggestion is somewhat similar to the pollution control system within the industrial estates described in Chapter 21.
- 6 Thanks to Anders Ekbohm and Wilfred Nyangena for interesting comments.
- 7 The relationship can go either way (Binswanger and Holden 1998). Population increase may speed up technical progress and lead to increased incomes. In many parts of Africa, however, where population increase is coupled with poorly developed institutions, development appears to have been more Malthusian.
- 8 India has similar schemes, such as the Self-Employed Women's Association (SEWA). Another successful scheme is the Aga Khan Rural Support Program in Pakistan. The Deutsche Bank runs, as a "proactive" commitment, a microcredit scheme that tries to help set up similar schemes in various countries. And participatory online microcredit services such as Kiva.org have become popular worldwide in the past decade.
- 9 This section builds on Vincent and Ali 2005, Khalid and Braden 1993, and Khalid and Wan Ali 1992.
- 10 The charge formula (Equation 26-1) was further modified by a required minimum that was not binding in the first couple of years, when effluent payments were sizable. However, the minimum payment would apply to a medium-sized firm when the effluents fell below 167 ppm, and technically, there would be no incentive below that level (Vincent and Ali 2005).

Forestry

Each year about 10 million hectares of forest is cleared in the tropics, and even larger forest areas may be subject to degradation. Tropical forests may be more than 150 million years old, so each hectare of forest has developed an immense and unique biodiversity that may be larger than that of enormous areas that have developed only since the last Ice Age (e.g., the northern hemisphere). Deforestation is not only a threat to biodiversity and local climate and water regulation, but also to climate change since it leads to large emissions of carbon dioxide (CO_2). Ironically, forests are increasing at the expense of agriculture in many high-income countries, such as the United States and Scandinavia, where high salaries and other factors favor this land use (see Chapter 22). This increase (at least partly) compensates for the loss in carbon due to tropical deforestation but does nothing to limit the loss in biodiversity or other local public good services that forests provide, such as climate stability.

This dichotomy has created a considerable although probably insufficient interest in the donor communities to protect forests, exemplified by the unexpected cooperation between the World Bank and World Wide Fund for Nature in a global alliance for forest conservation and sustainable use. Part of the interest in forest protection is related to carbon and climate change and part to biodiversity and other ecosystem services. This is not to say that all deforestation is inappropriate. It is impossible to define inappropriate deforestation because it is an inherently political issue (Kaimowitz and Angelsen 1998). However, deforestation is more likely to be inappropriate under certain circumstances, for example, on lands that

- are unsuitable for agriculture (due to soil quality or slope);
- provide valuable ecosystem services, such as protecting watersheds, or where the downstream effects of deforestation are high;
- contain considerable biodiversity; or
- are inhabited by forest users.

So many economic models explain deforestation that we can do no justice to the wealth of available ideas (see Kaimowitz and Angelsen 1998 for a survey).

However, one point that all the models agree on is the importance of access for increasing deforestation; this point is central to the model presented in Chapter 22. The effects of policy instruments or roads will vary depending on the type of forest. An economic analysis focuses on the costs of protecting ownership rights and the intensity of use and typically distinguishes between natural forests (areas where few people live and conventional agriculture has not yet developed) and managed stands (which may be densely populated, but the demand for wood has made commercial plantations more profitable than farming).

The economic interests behind public policies are also important. In many temperate forest countries, there has been a pervasive worry about resource depletion. It was partly a tactical concern for the wood processing industry, which was interested in access to cheap raw materials. Lately, however, it has been combined with more general ecological and social concerns. Forests perform several functions, ranging from nutrient recycling to the protection of watersheds, soil quality, and local climate. Clear-cutting in tropical forests can cause massive biodiversity losses, soil erosion, and the destruction of downstream ecosystems, including agriculture, dams, rivers, estuaries, and coral reefs. Such disturbances can have a serious detrimental effect on the local population, most significantly on the poor, who live on marginal lands and depend on ecosystem resources for water, firewood, game, and fishing.

Policy Instruments for Forestry

Various instruments are used in forestry management, including subsidies, taxes, and regulation of silvicultural practices. Three additional instruments have received attention in recent years: certification, international carbon offsets, and changes in property rights. Certification or labeling is a developed-country initiative designed to ensure that forest products come from sustainable forests, and carbon offsets are international payments to protect the carbon already sequestered in mature forests to help mitigate climate change. Ownership is important; the government is the largest forest landowner in most countries. Challenges to its effectiveness as a landowner during the past decade have led to greater focus on transferring some rights to (degraded) government forests to local communities and on contractual arrangements for the sale of government timber. The indirect effects on forests from agricultural instruments and roads can be substantial, particularly in developing countries. Certification, carbon rights, and ecotourism potentially offer the opportunity for policymaking in one country to affect forests in another, corresponding to two at least partly global goals: preserving biodiversity and mitigating climate change. In this chapter, we deal with each of these instruments in turn (except ecotourism, which is discussed in Chapter 28).

Subsidies

The main motivation for domestic forestry subsidies in the well-established forest nations such as Scandinavia and the United States was originally the fear of supply shortage (because the forest was needed for the navies, for heating, or for the timber and paper mill industries). The same fear also led to various other policy instruments, such as regulations for forest management. When subsidies are used, they tend to become permanent through the effects of the strong industry lobbies that benefit from them.

Most industrialized countries have some form of subsidies, often called forest incentives, which come in the form of free seedlings, management assistance, or financial aid for management. Financial aid is more common in developed countries that can afford it and where plantation forests are a large source of forest products. Even Chile's well-known free market has been modified with a financial aid program in forestry, and the success of Chile's forest sector leads many foresters in other countries to believe these subsidies are necessary for a successful forestry sector. Free seedlings and management assistance are common to forest policies and rural development programs around the world, regardless of a country's development status.

Despite the popularity of these instruments, response to them is sometimes limited. To understand the problem, refer back to the spatial model in Chapter 22. Incentives diminish management costs for professionally run plantations, but in the first stages of forest development—with large tracts of virgin forest—there are no management costs. Therefore, incentives begin to affect management behavior only during the third stage of forest development, when prices have risen so that landowners begin to manage their forests professionally. In this “mature” stage of forest development, incentives subsidize the cost of managed forests, raise the net forest value function, and extend both the intensive and extensive margins of managed forests. Still, forest incentives do not affect harvest or delivery costs and thus have no effect on extraction from the natural forest.

Developing country programs that have focused too exclusively on the number of trees planted or on the provision of seedlings have failed because they did not take into account the preferences (discount rates in particular) of local users (see Box 27-1). If the demand for agricultural products is higher than that for woody biomass, then free seedlings will have little effect on tree planting (and may hamper the domestic production of seedlings). In some settings, however, village plantations have used marginal lands that were of little use for agriculture and where indigenous tree cover would not easily have recovered. For example, some aid-financed village woodlots in the state of Orissa, India, significantly improved the welfare of poor communities (Köhlin 1998). These effects were due primarily to the time saved by collecting fuel close to home but also to the recovery of the natural forest when biomass collection pressure was reduced. A thorough understanding of local conditions

BOX 27-1 China's Sloping Land Conversion Program

The Sloping Land Conversion Program (SLCP) is one of China's most ambitious initiatives and one of the largest land-conservation programs in the developing world (Bennett 2008 and Xu et. al. 2010). Initiated in 1999, it has the purpose of reducing water and soil erosion and increasing China's forest cover by retiring steeply sloping and marginal lands from agricultural production. For that purpose, it provides subsidies to those farmers who convert degraded and highly sloping cropland back into either "ecological forests" (which in the Chinese context means timber-producing forests) or "economic forests" (plantations of trees with direct medical or other value). Farmers are compensated with an annual in kind subsidy of grain, a cash subsidy, and free seedlings provided at the beginning of the planting period (Xu et. al. 2010). The payments differ across regions to account for differences in average yields. In the case of ecological forests, the subsidies are provided during a period of eight years while the payments are limited to five years for economic forests.

The SLCP has expanded very quickly since its initiation; currently, the program engages millions of rural households. Although long-term success is still pending, it could increase the Chinese forest area by almost 10%. However, some problems in program design and implementation might risk this long-term goal and should be improved to increase the possibilities of program success. Among them, fast expansion has created some shortfalls in required funds, leading to problems in implementation and subsidy delivery.

On the other hand, the top-down approach has implied that some households have had unequal access to the program and, in some cases, have been forced to participate when they would otherwise have not. Voluntary participation would enhance cost-effectiveness by ensuring that households with the lowest opportunity costs participate.

Finally, Xu et al. (2010) find also evidence of mistargeting of plots for retirement in terms of the SLCP's stated target of highly sloping land, which indicates that considerations other than plot slope have been important in the enrollment choice of the villages.

is required to plan and site village woodlots properly and thus maximize their benefits. For instance, in general, village woodlots should not be located in the villages that border the natural forest but in those that are more distant, where people would otherwise have the farthest distance to travel.

Other developing-country subsidies that have attracted much attention are "perverse" subsidies that encourage deforestation (e.g., the famous Binswanger 1991 study on subsidies for forest destruction in Brazil mentioned in Chapter 7). Many criteria should be considered when selecting instruments to abolish subsidies on agricultural inputs. Such instruments include settlement schemes, agricultural extensions, taxes, and credit subsidies for farms on newly cleared land, logging industries, and roads. Policies will vary in their effectiveness, targetability, costs (direct and indirect), equity issues, and political viability (Kaimowitz, Byron, and Sunderlin 1998). Subsidy reduction meets resistance from powerful beneficiaries but also has the undeniable advantage of not spending but rather saving funds for the public budgets.

Taxes

Taxation is used mainly on private forestlands, to catch a share of scarcity or land rent (much like a property tax or taxes on other forms of wealth and income) or to avoid or correct for externalities created. The effect and efficiency of taxation depend on exactly how the tax base is defined, the type of forest, and the type of owner. Three common tax bases are land, standing timber, and harvested timber (the taxes for which are known as severance taxes). In addition, some communities have introduced preferential tax treatment for ecologically sensitive forestry (see Klemperer 1996 for a summary).

Ad valorem property taxes on forestland are generally expected to be neutral with respect to land use and rotation age (harvest timing). However, these taxes tend to accrue annually, and (at least small) managers obtain forest revenues with infrequent timber harvests. The timing mismatch between tax payments and forest revenues might create an incentive to convert the land to other uses.¹ Severance taxes on in situ resource value are also called stumpage taxes. In developing countries with valuable tropical hardwoods, the information asymmetries and monitoring difficulties make the collection of stumpage taxes difficult (Robinson et al. 2010). Studies have shown that stumpage fees are often so low that the resources are practically given away. In India, Bangladesh, and other Asian countries, large forest industries have paid only a tiny fraction of resource rents, thus in practice benefiting from a form of hidden subsidy (FAO 1993). The Hindustan Paper Corporation in Kerala, India, for instance, paid US\$0.50/ton on eucalyptus pulpwood when the Forest Department had direct production costs of almost US\$25/ton. Stumpage taxes could conceivably (if poorly designed) encourage highgrading (i.e., extracting only valuable tree species, to the detriment of other trees). Rational loggers leave more and more residual product in the logged forest as they move closer to the margin. It pays to remove only those logs that are worth more than their marginal costs, and because marginal costs are increasing as logging activity advances into the standing forest, they constitute a smaller and smaller share of the total forest. However, it may be possible to use differentiated stumpage taxes to specifically target highgrading (Amacher, Brazee, and Witvliet 2001).

One analysis of export taxes (and bans) indicates that these instruments may be moderately successful in reducing the logging of natural forests, particularly in the short run (Aylward et al. 1994). In the long run, their effects are less certain. The risk is that they reduce the value of timber and thus discourage serious investments in sustainable management, logging, and processing. This effect might encourage domestic consumption, which might lead to even greater logging (as is thought to have occurred in Indonesia). In principle, importing nations might impose bans or taxes with similar effects. However, this approach requires a fairly large number of countries and raises problems with trade politics.

Regulations

Forest regulation is common (particularly in temperate country forestry) and often pertains to the regeneration of private lands to ensure rapid reforestation after harvests of the previous stand of mature timber. Regulations initially were imposed in many countries to protect against anticipated timber supply “shortfalls,” which usually are not actual shortfalls but an argument used in the price bargaining between forest owners and sawmills or paper industries. However, reforestation regulations also may be motivated by environmental concerns for birds or biodiversity. Recently, some countries (including Sweden) have started to include specifically environmental objectives such as leaving dead trees for birds or leaving protective corridors along streams as a part of overall forestry regulations.

The effect of the regulations depends on the type of forest. For harvests of mature natural forest, requirements may delay harvests by adding a reforestation cost to the harvest cost. The reforestation requirements are an expression of the notion that the discount rate would be too high to warrant the regeneration costs.

It often comes as a surprise to people from tropical countries that forest owners in cold countries like Sweden plant trees despite a rotation period of a hundred years. This practice is partly due to regulations but also is ingrained into the local culture of private forest owners. In the United States, reforestation requirements have had no impact on the long-term standing timber volume on the largely private forestlands of Virginia (Boyd and Hyde 1989). In tropical countries, this form of reforestation regulation is uncommon. Other regulations may typically be included as part of the concession agreements. The “allowable cut,” for instance, is a common policy of most forestry agencies around the world. It restricts harvests of mature timber to a fraction of their standing volume (Davis and Johnson 1987; Hyde 1980).

Forest Concessions and Timber Contracts

One management strategy in countries with large government-owned forests is the granting of forest concessions to private forestry companies. The essential issues for concession policy include securing satisfactory financial proceeds for the treasury, providing incentives for sustainable and efficient harvest methods, and avoiding illegal logging while maintaining the long-term condition of the basic resource. If concessionaires can limit access and competition from illegal loggers, then they should feel sufficient security to manage their concessions sustainably; however, this is not always the case (Kaimowitz, Byron, and Sunderlin 1998). Concessions typically have been allocated noncompetitively, in ways that are nontransparent and possibly corrupt, partly because of the gigantic scale of some of these transactions. In 1968, for instance, Thailand issued 500 timber concessions covering half of the country (FAO 1993).

Associated with an irresponsible handling of concessions is inevitably a short-sighted and rapacious industry. Public forestlands are often either degraded open-access areas or inaccessible natural forest. In pure open-access areas, forest rents normally would already have been extracted, and hardly any would remain. In reality, the extent or ease of access depends on both technical variables (e.g., road quality, which decides logging and transport costs) and socioeconomic ones (e.g., regulations and control effort). As these variables change, forest rents may be created. Major roads such as the Trans-Amazon Highway in Brazil create massive rents. In Indonesia, more than 20% of all concessions operative in 1992 had lost their rights to operate by 1996. The cause for nonrenewal may have been disinterest on the part of the concessionaire, or the concessionaire may have violated acceptable long-term management standards. In either case, it is a sign that rents probably have dissipated.

Most forest ministries charge a fee per unit of log for the right to harvest.² The government thereby collects a portion of the rent, and the government typically is tempted to increase this royalty. Doing so might be highly commendable but will decrease the profitability of (legal) logging. Logging will decline, and royalties will increase or decrease depending on the elasticity of the marginal cost function. Unfortunately, this elasticity is seldom estimated. Some developing or formerly planned economies charge almost nothing and thus make no public revenue at all. Lithuania, for example, probably would increase the social profitability of the sector if a small, simple volume-based charge were imposed (Deacon 1996). Royalties also increase the incentive for illegal logging and for highgrading (see the similar discussion as related to fisheries in Chapter 25).

Concern for the condition of the forest at the end of the concession has led some environmentalists to suggest a liability bond on the environmental performance of concessionaires (Ruzicka 1979; Paris and Ruzicka 1989; see also Chapter 8). In the ideal case, the concessionaires would submit a bond at the outset of the harvest agreement. The bond would be returned on demonstration of acceptable performance. If the bond is truly a guarantor of environmental performance, then it must be set high enough to compensate for all costs of returning the forest environment to an acceptable condition in the event of noncompliance. This means higher bonds, for example, for concessions on steep lands and erosive soils and lower bonds for concessions in less erosive regions. Differences in logging effects on other environmental conditions require similar attention, so an assessment of environmental risks and mitigation costs should be conducted at the outset of the concession agreement. However, assessments are rarely conducted.

Certification

Certification and green labeling might be useful instruments to counteract information asymmetries and give consumers information about the environmental sustainability of various forest practices (see Chapters 8 and 9);

its greatest support comes from environmentally oriented consumers. Two of the main certifying organizations are the Pan European Forest Certification Council and the Forest Stewardship Council (FSC), based in Oaxaca, Mexico. The FSC is an international nongovernmental organization, founded in 1993 through the association of environmental groups, timber trade, forestry, indigenous peoples' organizations, community forestry groups, and forest product certification organizations around the world. As of May 2010, the FSC had certified more than 125 million hectares of forest in 80 countries; of this total, more than 10 million hectares were in Sweden, 7 million hectares in Poland, and more than 13 million hectares in the United States. Among the developing countries, Brazil, Mexico, and Bolivia together had about 7.8 million hectares certified by the FSC, whereas Malaysia and Indonesia together had only 1.3 million hectares (FSC 2010).

There is interest in certification in Indonesia, but not through the FSC. Large producers are scrambling to obtain certification and thereby gain European market shares. The Indonesian Ecolabeling Institute (Lembaga Ekolabel Indonesia [LEI]) has developed its own certification program that will comply with the standards of the International Tropical Timber Organization (ITTO), which "facilitates discussion, consultation, and international cooperation on issues relating to the international trade and utilization of tropical timber and the sustainable management of its resource base" (ITTO 2002) and will be monitored by an independent body. A protocol was signed in September 2000 between LEI and the FSC to jointly certify wood products under both programs. However, less than 10% of Indonesia's annual harvest comes from plantations; most of Indonesia's timber comes from natural forests. Even if Indonesia's producers obtain certification, monitoring will be a serious problem (to ensure that wood from uncertified lands is not substituted for certified wood). Meanwhile, the small institute is preoccupied with large forestland managers while the smaller private operations (which are likely to be the initial respondents to higher price incentives for sustainable management) have been overlooked.

REDD and Other Forms of International Payment³

Carbon offsets are essentially international subsidies or financial incentives to help mitigate climate change through carbon storage in natural reservoirs (e.g., soils and forests). Deforestation is a significant source of climate gases (mainly carbon dioxide). For instance, total emissions from tropical deforestation amount to some 4.4 (+/- 2.0) Gtons of CO₂/yr, rivaling the contribution of the global transport sector to climate change, though uncertainties still abound (van der Werf et al. 2009). Reforestation, afforestation, and "avoided deforestation" are possible "carbon sinks" (i.e., physical means of reducing the carbon content of the atmosphere).

The perceived need for some form of early action has led to the development of various funds for carbon payments by the Carbon Fund, the Global Environment Facility (GEF), and even individual countries and firms. Developing countries can apply to receive funds by finding and designating a suitable area that “fits into” the windows of opportunity created by the sponsoring organizations. Although far from ideal, this instrument opens up interesting possibilities in some cases.

On the other hand, although in the current global climate regime there are no incentives for developing countries to reduce the emissions from deforestation and forest degradation (as developing countries do not face emission targets), since 2005 there has been an ongoing discussion about the possibility of including carbon offsets from reduced emissions from deforestation and forest degradation (REDD) in a future climate regime. Indeed, this topic has been at the center of international climate negotiations in the last few years, for a number of reasons:

- developing countries themselves have expressed a willingness to contribute to climate mitigation,
- possibly large corollary benefits from forest conservation might appear (e.g., biodiversity), and
- large emission reductions might be achieved at a very low cost—both the Intergovernmental Panel on Climate Change (IPCC) and the *Stern Review* find REDD to be one of the least expensive abatement options available.

The basic idea—compensating developing countries for reduction in the emissions from deforestation and forest degradation—is straightforward and simple, yet creating a REDD regime that is environmentally effective, cost efficient, and equitable is a big challenge. Difficulties are related to coverage, baseline, and sources of funding (Angelsen et al. 2009a; Angelsen et al. 2009b; Parker et al. 2009).

Although initially the discussion was restricted to incentives to reduce deforestation, as negotiations progressed the scope of the discussion broadened. In the draft conclusions that came out of the 2009 Copenhagen climate summit it was agreed that a mechanism should also include incentives for reduced forest degradation, as well as for forest conservation and enhancement of existing forest carbon stocks—collectively referred to as “REDD+”. The “plus” is meant to signify that a REDD+ regime will not only benefit countries with high rates of deforestation but will also create financial incentives for countries with low rates of deforestation to protect existing forests and even expand them.

The expansion from REDD to REDD+ was partly driven by the concern that if rewards were only given for reducing deforestation rates, and not maintaining existing forest stocks, countries with large tracts of forest but currently low levels of deforestation would have no incentives to participate in the

REDD regime. The result of this could be international leakage, where the REDD system would simply lead to a shift in deforestation pressures from countries participating to those that are not, drastically reducing the environmental effectiveness of the regime—and even creating very perverse incentives for countries to start deforesting in order to become eligible to reduce deforestation rates! However, the shift was also driven by countries such as India and China—countries that historically have cut most of their forests and presently are increasing their forest cover again—who envision potential remuneration in the development of REDD.

A key issue in a REDD system is the emission level under which countries could receive compensation for reduced deforestation. The choice of this level, often referred to as the reference level (RL), will affect not only the environmental effectiveness of the REDD regime, but also the incentives for tropical countries to participate in the scheme—as is illustrated in Figure 27-1.

Quite often it is assumed that the RL should equal business as usual or average historical emissions—that is, allowances would be granted for all emissions reductions achieved through reduced deforestation. In Figure 27-1, this would be equivalent to setting RL equal to zero. Although this would maximize the rents captured by forest nations, it minimizes the environmental effectiveness of the system. Indeed, this would reduce REDD to a pure

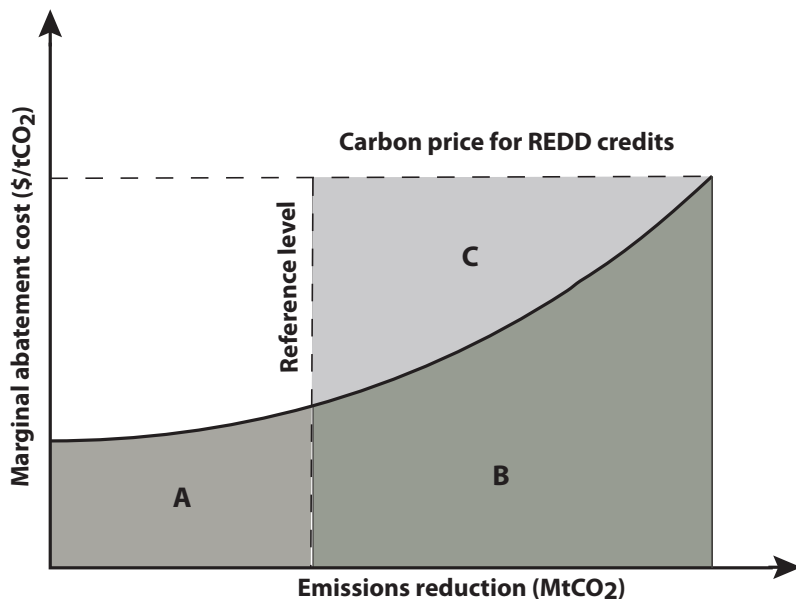


FIGURE 27-1 REDD and the Choice of Reference Level

Notes: Adapted from Angelsen (2008). The area A equals the total cost for emission reductions that is paid by the country itself, area B the abatement cost financed by the REDD system (either through the carbon market or a REDD fund), and area C the rents captured by the forest country.

offset scheme, not contributing to an overall reduction in emissions (since all credits will be sold to the market, leading to higher emissions elsewhere), only lowering the total cost for a certain global emission target. In a fund-based system it would mean getting the least amount of reduction in forest clearing for a given amount of funds.

Maximum environmental effectiveness—in both a market and fund-based system—would be achieved by setting the RL so that the REDD rents exactly offset the cost for emission reductions needed to reach the RL ($A = C$ in Figure 27-1). At this RL a forest nation should be indifferent between participating in the REDD system or not, since the net benefit for them is zero. There is thus a trade-off between environmental effectiveness on the one hand, and incentives for participation and benefit sharing on the other.

An additional complexity to the issue of setting RLs is the fact that deforestation rates are highly variable and respond to changes in the underlying drivers of forest clearing such as international prices for timber or agricultural produce like soybeans, palm oil, or beef. Predicting future deforestation rates is therefore extremely difficult. If RLs are too lax we risk admitting credit emission reductions that are not additional—i.e., that would have occurred anyway (Persson and Azar 2007).

For example, consider the fact that deforestation rates in the Brazilian Amazon dropped by more than 60% in the period 2005–2009 (Nepstad et al. 2009). The amount of associated emission reductions the Brazilian government should be credited for depends on how much of the decline in clearing is a result of policies (e.g., crackdowns on illegal logging and expansion of protected areas) and how much was merely a result of reduced profitability in the soy and beef industries. Finally, if the basic idea with REDD is that developing countries should be economically compensated for reducing deforestation, and hence carbon emissions, a key question is of course where the money to finance the system should come from. Today it is increasingly recognized that funding will have to come from a variety of sources, both private (carbon markets) and public (bilateral and multilateral funds) in what is usually referred to as a phased approach.

In the early phases of REDD, where the main focus will be on building institutional capacity for monitoring land use change, assessing its main drivers, and developing policies for controlling it, the primary source of financing will likely be through international funds. In later stages, when the effectiveness of enacted policies can be monitored and verified and performance-based payments can be made, the feasibility of tapping into the global carbon market is larger.

How successfully REDD will arrest deforestation and forest degradation depends in the end on the policies adopted nationally to implement REDD+ and how REDD+ payments will compete with other land use options. While hundreds of REDD pilot projects have already been launched, and large donors have started different REDD+ initiatives (e.g., the United Nation's UN-REDD

program and the World Bank's Forest Carbon Partnership Facility), it has been argued that "most planned national policies to be implemented are comparable to measures tried in the past—often with disappointing outcomes." (Angelsen et al. 2009a, *xiii*). If REDD+ policies are to have an effect they must learn from previous attempts to reduce deforestation that failed. They must succeed in building political support for forest conservation in tropical forest countries (Sunderlin and Atmadja 2009).

REDD may offer an incentive for reduced deforestation by pricing carbon emissions from forest clearing. Nevertheless, climate policy—through a carbon tax or a cap-and-trade system—will also augment the willingness to pay for bioenergy, raising the demand and price of agricultural land and hence the profitability of deforestation. This implies that REDD payments may not suffice to make forest clearing for high yielding bioenergy plantation like palm oil unprofitable (Persson and Azar 2009). This of course calls into question the hypothesis on which REDD rests: that these credits will be cheap. In fact, additional, and stronger, protection measures for the world's tropical forests may well still be needed.

Clarification of Property Rights

In most countries, the state is the largest single forest landowner and at some time in the past it has declared itself the ultimate owner of all unclaimed resources. Much of the national forest was originally land "left over" after private claims were established. On large tracts of prime forest, the government commonly grants forest concessions; other areas of public forests are either degraded or inaccessible, so the management is expensive relative to the production values at stake and appears inefficient. The state may be an unsuccessful forest manager for other reasons. The two most common criticisms are that the state fails to incorporate local demands on the forest and does a poor job of collecting revenues. In addition, government employees sometimes engage in rent-seeking.

The state is effectively an absentee landowner. Many of its problems are related to the vast size of government holdings (see Chapter 3). It does not have the same knowledge and control of the land that a local manager would, even if a ranger is stationed on the land, and often manages the forest for timber regardless of local land use preferences for a multitude of goods and services. Consequently, local users typically trespass to extract whatever they can, treating the forest as an open-access resource even if there is some level of government protection of its property rights.

If forests are already degraded and the government agency is frustrated with its own attempts at management, some or all property rights might be shared with local users. Examples include Panchayat Forestry in Nepal (see Box 27-2), Joint Forest Management in India, community-based forest

BOX 27-2 Forestry and the Poor: User Rights and Afforestation

The relationship between poverty and deforestation—and the role of gender—are much debated, complex issues. The view that the poor (in their quest for land) are a main cause of deforestation is now widely challenged. Very poor, landless populations are more than proportionately dependent on open-access or common property resources. They suffer most when firewood, medicinal plants, game, nuts, fruit, and other goods become scarce. Women and children are particularly affected because collecting water and firewood is often their task.

In recognition of this, some policymakers have started to involve local communities in forestry programs. In 1978, the Nepali government passed legislation enabling public lands in the middle hills to be handed to the local community organizations, *panchayats*. They were very bureaucratic at first but eventually decentralized all decisionmaking to user groups. In June 1997, 6,000 small, informal but autonomous user groups managed 450,000 hectares (Shrestha 1996). In a country that has witnessed drastic deforestation, these user groups appear to be improving the condition of the forests.

management in the Philippines, and community forestry in many developing countries. The effect of the transfer of rights is to decrease the cost of protecting land use rights and thus increase forest cover because local management is generally less expensive. Successful examples seem rare but a few from India illustrate the conditions for successful joint community–government forest management (Kant 1996).

If state ownership is inefficient and there is some hesitancy about large private companies, then small-scale private ownership may appear attractive. Many people view such an ownership pattern as hypothetical, but it is common in Europe, particularly in Scandinavia, where a significant share of forests are owned and managed by (small-scale) private owners. When this kind of small private forest owner does not exist, common property resource (CPR) management may be the most adequate solution, particularly for lands that have low and strongly variable productivity (Ostrom 1990). Community control, together with appropriate policy instruments such as user fees (wood taxes), may encourage the sustainable recovery of degraded open-access lands in the Sahel region (Chomitz and Griffiths 2001). The World Bank has abandoned its earlier pro-privatization policy—as expressed in the 1975 Land Reform Policy Paper (Deininger and Binswanger 1999)—and now favors communal tenure systems. The bank recognizes that communal tenure may be more cost-effective than formal titles and that, with imperfect credit markets, titling programs may favor the better off, lead to land concentration, and sometimes cause inequity and inefficiency.

Supplemental Reading

Alvarez and Koskela 2007	Gibson et al. 2005
Barbier and Burgess 1997	Hyde and Amacher 2000
Burgess and Barbier 1999	Larsen and Nielsen 2007
Coleman 2009	Mekonnen 1998
Galinato and Uchida 2010	Mendelsohn 1994

Notes

- 1 In this case, the taxes would not be neutral. However, in Sweden, the tax authorities allow forest owners special accounts to even out their flow of income. Ad valorem taxes on land and standing timber may discriminate against forestry because they tax both stock and growth every year until harvest, which could create an incentive for earlier-than-optimal harvesting. A separate problem is that small forest owners (particularly part-time foresters) might fail to react to tax incentives because they lack information.
- 2 To maximize the rent, it probably would be better to conduct an auction. Some forestry transactions between private landowners and loggers or mills are conducted this way (Hyde and Sedjo 1992).
- 3 Many thanks to Martin Persson in this section.

Ecosystems

Natural resources such as forests and fisheries are complex. Yet ecosystems are much more complicated than the models discussed in previous chapters in Part VI. Managed ecosystems called “agriculture,” “aquaculture,” and “silviculture” are purposefully simplified for ease of management and for human enjoyment. The natural ecosystems (on which managed systems are inevitably dependent) are several orders of magnitude more complex than managed systems; many vital functions are derived from them, and thus the future of the human race depends on preserving them (see Chapter 22). Chapter 26 touched on this dependence in the discussion of wetland restoration.

Humankind’s failure to foresee these complexities can be illustrated in anecdotes like the following:

Ernst Mayr tells of a steamer wrecked off Lord Howe Island east of Australia in 1918. Its rats swam ashore. In two years they had so nearly exterminated the native birds that an islander wrote, “This paradise of birds has become a wilderness, and the quietness of death reigns where all was melody.” On Tristan da Cunha almost all of the unique land birds that had evolved there in the course of the ages were exterminated by hogs and rats. The native fauna of the island of Tahiti is losing ground against the horde of alien species that man has introduced. The Hawaiian Islands, which have lost their native plants and animals faster than almost any other area in the world, are a classic example of the results of interfering with natural balances. (Carson 1950, 78)

In general, the introduction of exotic species—an important element in managed agriculture—introduces numerous risks of this type. Consider the water hyacinth, introduced as an ornamental plant, and the Nile perch, which together have completely transformed the ecology of Lake Victoria, reducing biodiversity considerably. The perch has outcompeted or preyed on many of the native species, including a multitude of endemic cichlids, and the hyacinth has spread invasively (see Chapter 26). With modern forms of communication

and technology such as genetic engineering, the risk of spreading alien species and disturbances to natural ecosystems may increase drastically.

One of the essential inherent characteristics of many natural ecosystems, particularly the more sensitive ones, is high variability. This variability interacts in a complex way with social variability. On the one hand, people adapt, so culture and social organization reflect the underlying natural conditions. On the other hand, people exert a considerable influence on their landscapes—sometimes improving their productivity and beauty, sometimes degrading them instead¹.

In this chapter, we illustrate policy ideas presented elsewhere in this book with examples of ecosystem management. Although negative examples and alarming signals abound in this area, we emphasize selected positive (and, we hope, inspiring) examples of careful and perhaps sustainable natural resource management. Various instruments are particularly suitable for ecosystem protection: forms of zoning; the creation, maintenance, and management of parks; and common property resource (CPR) or community-based range management, particularly wildlife management. We will look specifically at the Communal Areas Management Programme for Indigenous Resources (CAMPFIRE) in Zimbabwe, coastal zone protection for coral reefs off of Tanzania and Mexico, and direct payments for ecosystem services in Costa Rica.

CPR Management of Wildlife in Zimbabwe

The crucial importance of appropriate ownership rights and empowering local communities to resolve conflicts between farming and wildlife are well illustrated in CAMPFIRE.² Started in 1989, the program has been run with some funding by donors but is also generating its own funds—primarily from big trophy hunting—and there is hope that it will find a way to a sustainable and independent existence. This project also illustrates the severe crisis that has resulted from pursued policies that exacerbate great political tensions and declining investments that follow from a lack of confidence in the regime. Under such circumstances, even a well-designed project such as CAMPFIRE that is dependent on hunters who are mainly expatriates is bound to suffer.

In Zimbabwe, as in many other areas, the legacy of colonialism was destructive. All wildlife in Rhodesia was classified as the property of the British crown (“king’s game”), and local hunting was banned. Sometimes, even the local inhabitants were removed so as not to disturb the gentlemen’s hunting. Local inhabitants often thought of wildlife as the symbol of oppression and “poachers” as heroes, particularly if they ridded the community of animals that were dangerous or competed with livestock for pasture³.

In 1975, the Wildlife Department implemented the Parks and Wildlife Act, which gave private landowners “appropriate authority.” The purpose of this legislation was to allow and in fact encourage landowners to manage wildlife,

including any commercial use they saw fit. The hard-line conservationists (who wanted to ban all commercial exploitation of wild animals) were skeptical but soon found that commercial ranchers started to protect elephants and other animals rather than kill them. Several studies have shown that in some dry areas, wildlife ranching is much more economically profitable than traditional cattle ranching (e.g., Bond 1993; Jansen, Bond, and Child 1992; Skonhofs 1998). The reasons are largely ecological: the local vegetation—shrub and bushes—is not good pasture for cattle, which fall prey to parasites and predators. These problems lead to increased costs for the ranchers and less-than-optimal growth of the cattle. Wildlife is better adapted to the local ecology, with which it has coevolved. Browsing patterns, resistance to disease, and many other characteristics ensure that the wildlife survive better than cattle. They may give less meat, but if that meat fetches a higher price and if money can be made from trophies, hides, hunting, and tourism, then the overall profitability may be higher.

From 1978 to 1986, the Windfall program was implemented, which had only a weak link between wildlife and remuneration and benefited mainly the large farms. In the next phase, starting in 1989, CAMPFIRE extended the devolution of appropriate authority to local communities throughout most of the country. It was a significant step, because small-scale farmers in simple housing bear most of the risk and costs of living with wildlife. Therefore, the resistance to wildlife is strongest in these groups. Even in the 1980s, poaching was still common, and a dozen poachers were wounded, captured, or killed each year by rangers.

For diversified wildlife to thrive, they need not only protection from poaching but more active policies that ensure an appropriate environment. Such policies affect the erection of fences and road construction, pesticide use, and many other aspects of farming and infrastructure planning. Also, human infrastructures need to be built differently to accommodate the inevitable pressure that some types of wildlife represent. For all this to happen, local communities must share in the benefits and must be empowered to make decisions.

To encourage the coexistence of farming with wildlife, CAMPFIRE has built organizations at various levels. This organization parallels the administrative organization but at the same time is its mirror image. CAMPFIRE organizations exist at the local, regional, and national levels. The smallest decisionmaking unit is the village, which typically consists of 100–200 households, even though the CAMPFIRE ward (which typically consists of some 1,000 households, or up to 10 villages) has been more active. On average, 25 wards (10,000–50,000 households) make up a district, and the district council consists of all the traditional chiefs in the area as well as an elected representative from each ward. The Rural District Council (RDC) decides how much money to pass on to the wards or villages. Typically, 20% is retained by the RDC and 80% passed on to the subdistrict units. Of the money retained by the RDC, 15% is earmarked for managing CAMPFIRE in the district and 5% for

general council administration and development. The allocation of revenues initially between wards and subsequently between villages is a central administrative issue, and conflicts have arisen over distribution principles between wards, between villages, and between the elected CAMPFIRE representatives and the traditional chiefs.

Nongovernmental organizations (NGOs) have a role to play, mainly in an advisory capacity. CAMPFIRE as an organization has characteristics of a CPR institution, an NGO, and a government organization. One is reminded in this context of Ostrom's seven principles for successful CPR management (Ostrom 1990; see also Chapter 8). Rule 3 recommends democratic and participatory decisionmaking, whereas Rule 7 requires respect for the CPR institutions by higher levels of government, and Rule 4 requires local monitoring. These requirements are only partially fulfilled in CAMPFIRE, and additional development would require a greater devolution of appropriate authority. Rule 2 concerns the importance of adapting provision and appropriation rules to the local ecological and sociocultural conditions. It is difficult to create a working community when the borders are imposed by government arbitrarily (Muchapondwa 2001). The CAMPFIRE districts are modeled on the administrative borders within the country, and the borders between districts often have been drawn without regard for ecology, wildlife concentration, or sociocultural structures.

An alternative being considered in Namibia (where there are plans for a somewhat similar wildlife management program) is to let local communities define themselves as districts. This approach may cause difficulties too, but it would have the advantage of creating local ownership and legitimacy.

Popular Participation in Chikwarakwara

A project such as CAMPFIRE needs "success stories," partly as a pedagogical tool. The following is such a story concerning the build-up of appropriate institutions.

A community in Chikwarakwara, a Zimbabwean village near the border of Kruger National Park, South Africa, has managed democratic decision-making—including the distribution of benefits, decisions concerning wildlife, and public goods—in a context where few people can read (Child, Ward, and Tavengwa 1997). In 1989, the district to which Chikwarakwara belongs sold permits to hunt three elephants, some buffalo, and other animals and decided that the revenues should benefit the community where the animals were shot. Because Chikwarakwara had much wildlife, several animals were shot there, and the village earned a large sum of money (60,000 Zimbabwe dollars [Z\$]). The whole community gathered for four days beneath a baobab tree, and the money earned from trophy hunting and other sources was piled up in large piles of common notes. The exact sources of the money were carefully accounted for as a way of emphasizing the value of the various wild animals.

The next step was to establish a list of village membership. Then, a long, open discussion took place about how to use the money: as cash distributions to village residents, or as funding for common (public good) projects. The almost unanimous decision was to spend half the money on each: thus, each of the 149 households would receive Z\$200 in cash, and the equivalent of Z\$200 per household would fund a grinding mill and the school. (Characteristically enough, some young men wanted a larger individual cash share).

To legitimize decisions and emphasize the participatory nature of the decision, a final payment ritual was carried out. Each household came forward separately, received its total share (Z\$400) in notes, then paid for the public goods that had been agreed upon by splitting its Z\$200 contribution into two separate bowls. This unusually transparent procedure was designed to involve every member of the community and avoid corruption and the temptation (and the opportunity) to free ride as well as the hesitancy in decisionmaking and other inefficiencies that accompany even the suspicion of such corruption. The grinding mill was built within three months and has proved to be a catalyst for the village. It is so well managed that it makes a profit (which is unusual for council-run mills). The village has started to strengthen its communal planning in other areas, looking over its water supply, applying for donor funding for a rotating fund to finance the purchase of pumps, limiting cattle numbers, and employing a villager as game warden to protect wildlife.

Looking Ahead

What happened in Chikwarakwara is probably an ideal seldom attained, and it may be an exception to the rule. In its first decade, CAMPFIRE showed some important signs of success but also some considerable difficulties due to a very difficult sociopolitical environment. Before CAMPFIRE, some 50 trophy bulls were shot each year by safari hunters and 200–300 as “problem animals.” Now farmers and communities think twice about classifying animals this way because they can make so much money from hunting fees: in the second half of the 1990s, the number of problem animals shot had dropped to around 30 per year, whereas around 90 trophy bulls were shot each year at a value of up to US\$10,000 apiece. From 1989 to 1995, CAMPFIRE income grew from US\$300,000 to US\$1.5 million per year only in the 12 original districts that first started to implement the program, which later spread to most of the other districts in Zimbabwe. Aside from the financial and ecological benefits, the program gave the communities valuable experience in marketing, tourism, and negotiation with tour operators. The communities have received increasing shares as their negotiating skills have increased.

Even though CAMPFIRE is a promising program, it faces several threats: conservationists want to save elephants by prohibiting all hunting and by stopping international trade in trophies, hides, and ivory. Since 1999, the political turmoil concerning land distribution and elections that has surfaced scared

away hunters, tourists, and other investors, thus presenting a new threat not only to CAMPFIRE but to the Zimbabwean economy as a whole. Since around 2005, the country has experienced dramatic hyperinflation and stagnation, and President Mugabe has lost more and more legitimacy. CAMPFIRE has still shown that the best way to protect wildlife appears to be giving appropriate property rights to local communities. At this general level there is broad agreement, but Fischer et al. (2011) show that actually designing optimal instruments is quite complicated. Their bioeconomic model—measuring growth in wildlife but also behavioral functions for poachers, farmers, and wildlife agencies—shows that it is not enough to share the profits from hunting and tourism; it is also vital to allow the communities to have a say in the quota setting process, which is where the key to controlling wildlife stocks lies. If the local communities do not expect the parks agency to change hunting quotas in response to a larger wildlife stock, their additional efforts to resist poaching and promote conservation may be undone completely, since they will be very averse to wildlife stocks that are so big they pose significant threats to local agriculture and subsistence.

Willingness to Pay and to Visit Parks

One of the most emblematic of all nature conservancy methods is the creation of national parks. They have, however, often been controversial for a number of reasons. Parks take land away from agriculture, forestry, or other activities, and on more than a few occasions local inhabitants have been quite forcefully displaced by park creation. Furthermore, national parks require quite sizable budgets to maintain and monitor.

The case is made that national parks are very important for biodiversity, ecosystem services, option and existence values, and the more concrete: They attract tourists. But what have the parks actually got to prove all this? Critics who want to see revenue note that revenue from parks has often been low. The main reason for this is often that fees are low—partly for reasons of social justice, because if they were higher, local residents with low income would be excluded. In recent years, however, both a considerable amount of research and a change in practical policies has taken place. The Environment for Development centers in Costa Rica and South Africa have taken a lead in this development.⁴

Many developing states, however, are seeing an increasing trend of phasing out tax-based grants to national parks, placing even more pressure on them to set prices that will cover their costs. The pricing of national parks is also very important for at least two other reasons: first, it can be used to manage congestion, and second, it can be used to smooth seasonality in park visitation. Finally price discriminating policies should take into account the different income levels of national and international visitors—and maybe also other categories, such as students or pensioners. Alpízar (2006) develops a model

which allows for differential pricing of visitors to national parks based on Ramsey pricing. More recently Alpízar and colleagues have collaborated with the parks authorities in Costa Rica to test how revenues can be maximized by making contributions voluntary (partially) and providing information, gifts, or other incentives to elicit higher payments (Alpízar et al. 2008a; 2008b).

Protection of Marine Ecosystems

People typically think that nature conservation is a task for national governments, and that parks are places you walk or drive in. This section⁵ examines some marine parks—a couple that are privately run and one that is managed as a CPR—in an attempt to illustrate that many forms of management may be appropriate, depending on the circumstances.

The earliest examples of protected areas were private initiatives, created for recreation exclusively enjoyed by a small group of people.⁶ The end result was that some important areas were preserved for posterity. However, this kind of conservation created considerable animosity, threatening its sustainability, because it was founded on the exclusion of poor people. Public policy is needed to manage the many market failures in ecosystem “provision”; however, the public sector must not necessarily be the manager. In practice, the state may seem to be a natural candidate for manager; take, for instance, the 200-mile exclusive zone of coastal states, or the Antarctic, which appears to be reasonably well protected by several nation-states. However, public authorities actually can be clumsy managers, which is one reason why even traditional public enterprises such as railways or telecommunications are privatized. Many public parks are poorly managed (Braatz 1992), and cooperative or private management should not be excluded. Many developing countries—South Africa, Namibia, Botswana, Costa Rica, and Kenya to name a few—have privately protected areas. In some regions, they are larger than the publicly managed parks (Watkins, Barrett, and Paine 1996; see also Langholz et al. 2000).

Many coastal zones in the tropics containing striking biodiversity perform vital ecosystem functions, such as providing food and protection against erosion for densely populated regions. Developments in medicine have created a greater awareness of the potential for new biologically potent compounds from these environments. Yet protected marine areas are a new and perhaps controversial concept. The traditional absence of clearly defined property rights—not to mention the technical difficulties of implementing any such rights because of the nonterritorial or migratory nature of many aquatic resources—make the delineation of a park problematic. Management issues such as exclusion, enforcement, and monitoring pose special problems that do not exist for terrestrial parks. Cheap airfares and high incomes in industrialized countries are creating a growing tourism industry that is increasingly sophisticated. The development of safer and cheaper diving gear and underwater

photography equipment is creating a greater interest in healthy, untouched reefs (Davis and Tisdell 1996). The concept behind marine ecotourism is to create economically viable tourism that provides employment and income to the local community by protecting the natural resources on which the community depends in the long run.

Mafia and Zanzibar

The case of protected marine areas off the coast of Tanzania is not very encouraging. In 1975, under the country's Fisheries Act, eight areas were declared marine reserves. Most of these areas have remained "parks" on paper only, and their condition has deteriorated so much that they are hardly worth protecting any longer. One, Maziwi Island, has even disappeared as a result of erosion, probably caused by damage to reefs (Fay 1992).

Mafia, the park with the most local CPR participation, has been most successful—perhaps because it is not exposed to heavy exploitation (it is farthest from the mainland). Its exceptionally pristine marine ecosystems have attracted considerable attention from the scientific, donor, and conservation communities, leading to the formal protection of Mafia in the Marine Parks and Reserves Act in 1994, a major achievement. Proper representation of all the interested parties—including different groups of fishermen and local cooperative involvement—appears to be key to potential success in Mafia, but progress has been slow, the cooperative approach has its limitations (especially in areas with little population), and much remains to be done (Andersson and Ngazi 1995).

Next to the main island of Zanzibar are two small islands of special interest: Mnemba and Chumbe. Mnemba Island is home to an extremely luxurious exclusive resort (Bill Gates has been there) off the north coast. The island is carefully protected from intruders, which means that the income-generating activities of collecting seashells, octopus, and sea cucumber are no longer available to the local people. The owners presumably take care to protect the local environment for the sake of their wealthy customers, but there is no evidence that they have used any advanced ecological concepts.

Chumbe Island became Zanzibar's first marine protected area in 1994. The project started as a private initiative in 1992 and is the result of one person's ambitions and perseverance. The project holds a 33-year-old lease of 2.5 hectares of cleared land and a management contract of 10 years for the Chumbe Reef Sanctuary. The site was identified as particularly suitable for its purpose: The marine environment possessed a rare example of pristine coral island ecosystems in an otherwise overfished and overexploited area. The island was uninhabited and the surrounding waters were not heavily used by fishermen, who might otherwise have felt that they were expelled from their fishing grounds. Thus, the risk for potential conflicts was minimal, and the small size of the island was particularly suitable for a privately managed marine park. The project design and management plan reflect great environmental

concern and awareness. And because the venture is privately financed, it must be financially viable in the long run.

The ultimate goals of the project are conservation (of the reef and island) and cultivating an understanding of the ecological importance of the reefs. An important part of the management plan is the construction of an environmental education program for local people. Five former fishermen have been employed and trained as park rangers, and school children regularly visit the island as part of an effort to increase understanding of this project. Although ecotourism is defined as an activity that can sustain and finance the other goals and make the project viable and independent in the long run, this result is doubtful, because the costs are driven up by high environmental ambitions. For the same reason, only limited numbers of tourists will be allowed, and thus earning capacity is limited.

The island is 16 hectares in area, so its carrying capacity for ecotourism is limited. Chumbe had no freshwater facilities or electricity, and because coral reefs are extremely sensitive to nutrients, the issue of sanitary and other waste was central to the planning of the venture. The solution to these challenges has been the use of sophisticated ecotechnology—state-of-the-art in some countries but definitely advanced by Tanzanian standards. Seven bungalows, housing a total of 14 people, are each equipped with a composting toilet; rain-water is collected and stored for use in the showers. Graywater is filtered in sand beds and used to irrigate gardens. Each bungalow has solar heating for water and photovoltaic cells that charge batteries for nighttime lighting.

Many other details of the island's infrastructure such as footpaths and project buildings reflect the same careful ecological thinking. This effort is appreciated by some people but also, perhaps ironically, has been a source of various problems. The technology is expensive, and the style is not easily recognized by local authorities, who sometimes perceive it as more eccentric than useful. The project has had difficulty being officially recognized as a luxury hotel because the usual attributes of luxury hotels—bars, swimming pools, discos, and even a multistory cement dwelling—are all missing. Resulting uncertainty and resistance from authorities causes additional delays and increased project costs. In fact, one option that would have been attractive to the project was to leave the island completely undeveloped and make only day excursions. However, this option was not possible, because a requirement of the government lease was the construction of hotel facilities or similar development.

With high costs, another dilemma surfaces concerning pricing: To recover the high costs of this ambitious project with only a limited number of beds, prices would have to be high. Such prices require sophisticated marketing and also may entail some restrictions—for instance, on the acceptability to the customers of daytime tourists and school children. These issues illustrate the difficulty and potential clash of interests among ecological awareness, interaction with the local community, and the interests of clients prepared to pay high rates and demand exclusive treatment.

The experience of Chumbe shows that, in some cases, private protection can be a viable and desirable way to protect a marine ecosystem. One important issue is the site features. Chumbe is a manageable size for a private entrepreneur but would be very small for a public agency. Another feature that made Chumbe ideal is how little the site had been used by the local people. Furthermore, the ecological characteristics of a site determine how successful protection will be in producing ecosystem services. Chumbe appears to have the potential to be an ecologically important retreat and recruitment area for various corals and fish species to replenish the reefs near Zanzibar Island (compare the role of marine sanctuaries in metapopulation models discussed in Chapter 22).

A protected area that produces external benefits or public good services may need some form of compensation—not necessarily financial, but perhaps simplified licensing procedures. A small operator has limited handling capacity for bureaucratic issues. Development projects are most successful when the policy environment allows frictionless and secure operations (Lewis 1988). Without the adequate legal infrastructure, ownership security, and good governmental relations, the transaction costs might become prohibitive. Former socialist countries such as Tanzania still harbor some “antiprivate” attitudes. Among the institutional prerequisites for private protection, the most important would be the security of the ownership. Without it, a private enterprise tends to set up short-term objectives at the expense of long-term sustainability.

There is no single “best” management structure for a national marine park, but many models should be tried and evaluated. Private protection probably would not be the only or even the main method for protection. Important stakeholders should be involved, and CPR schemes might be preferable. However, different parties can learn from each other and collaborate to improve managerial efficiency and to develop ecological engineering and resource-extraction techniques.

In the Tanzanian cases, Mnemba Island has profitability with fairly limited ecological ambitions; Chumbe Island has considerable ecological credibility but is hardly commercially viable; Mafia, which represents a much larger and more complex set of islands, has stakeholder participation but numerous problems concerning viability and sustainability. Initially, some of these projects will be better in technology and others in social conflict resolution. Some will be more ecological, and others may be more profitable—but heterogeneity and collaboration may give great benefits in this area.

Punta Nizuc, a “Sacrificial Reef” in Cancún, Mexico⁷

Close to the northern tip of the Yucatán Peninsula is the resort of Cancún. The beaches of Cancún draw approximately 2.5 million visitors a year, making it one of the most popular tourist destinations in the Caribbean. With a unique combination of archaeological sites, beautiful beaches, good food, pleasant weather, and proximity to the United States, Cancún provides an opportunity

for even further expansion of the tourist trade. However, tourism can “self-destruct” through a form of overexploitation in which the crowding and deterioration of sites lead to a vicious circle of lower expectations, lower pricing of hotels and tours, poor management, and further degradation. It is therefore crucial not only in terms of ecology but also commercial viability to coordinate the tourist industry in such a way as to maintain natural resources.

Restricting the number of tourists is typically the hardest and least popular option, but some other options can minimize the ecological damage they do: zoning, information and education, and technical solutions like providing buoys for mooring tourist boats. All these options require some way to fund park staff and services, which generally are provided through park fees.

Despite Cancún’s large-scale tourist operations and big hotels lining the beaches, the area contains a diverse and unique combination of coral reefs and marine life as a result of its location: in the Yucatán Channel, at the juncture of the Caribbean Sea with the Gulf of Mexico. It forms part of a coral reef system that extends to Belize and contains a considerable variety of fish species that thrive in the Caribbean Sea. Several islands, islets, coral reefs, and other features of potential interest to both ecologists and tourists include:

- Isla Mujeres, an island with a fully developed although low-key tourist industry of “family” hotels that typically attract European backpackers;
- Nichupté lagoon;
- Punta Nizuc at the south end of the Nichupté lagoon, just outside of which are readily accessible coral reefs in shallow and protected waters;
- Coral reefs such as the large Arrecife Islache and El Cabezo—relatively difficult to access but attracting trained scuba divers (other reefs are closer to Isla Mujeres);
- Isla Contoy, the most distant of the islands and uninhabited—declared a national park through several laws starting in 1961—and one of the few islands of the Mexican Caribbean whose ecosystems are still relatively natural.

In addition to the park on Isla Contoy, there are more parks such as Punta Cancún and Punta Nizuc (NMP). These two parks illustrate several important aspects about management, which we summarize in the categories of zoning, damage prevention, and funding.

Zoning

A growing city that receives millions of visitors each year will not pass unnoticed in the local ecology. Cancún can and should still aspire to have clean beaches and to minimize pollution, but it is impossible to keep the central areas in pristine condition. Although truly pristine conditions are not demanded by the average tourist, the number of people who want to spend at least a small part of their time looking at mangrove forests and coral reefs is increasing. In this situation,

which poses a somewhat paradoxical risk for the destruction of resources, zoning allows for advantageous policies. The Isla Contoy park, for example, is used mainly for research and for visits by those with a greater interest in reefs, bird watching, and so forth. Isla Contoy is farther away from the main Cancún tourist spots, and numerous visitor restrictions reduce the number of tourists.

Snorkeling, scuba diving, and “jungle tours” of mangroves are the main uses of the NMP. (The park also permits some commercial lobster fishing.) Even within the NMP, there is a high degree of specialization: Punta Nizuc is used almost exclusively for snorkeling, while most of the scuba diving takes place off Isla Mujeres (only a fraction of the numbers at Punta Nizuc) and Punta Cancún. Because Punta Nizuc receives the greatest flow of visitors, concern has been raised over its deterioration. However, Punta Nizuc has all the characteristics of a perfect “beginners’ and tourists’ reef”: a shallow area protected from waves, safe and ideal for snorkeling. Therefore, it is somehow destined to be sacrificed to mass tourism in order to protect other areas and keep them in a more pristine condition. The entire strip is heavily developed with hotels, and it is almost impossible to imagine it being pristine—but it can be kept attractive enough to generate a significant stream of economic benefits in the Cancún tourism industry.

Damage Management and Monitoring

The vast majority of visitors to NMP come on boat tours, which start inside the lagoon and thus are relatively easy to monitor. They use small two-person motorboats, with one private guide for every five boats, or larger boats that accommodate up to 30 people. Visitors are first taken on a tour of the mangroves inside the lagoon. Next, they are led through a narrow channel that passes under the Nizuc bridge and out into the Punta Nizuc marine park, where they are taken on a snorkeling tour of the coral reefs. This narrow channel is the principal access point to the snorkeling area and makes monitoring the boats fairly easy.

Park management is concerned with controlling the damages caused by visitors to the marine park and to Punta Nizuc in particular. The fear is that excessive damage may eventually lead to the ecological and economic collapse of the park. The main sources of damages are:

- physical contact between visitors and the reef, including taking pieces of coral as souvenirs;
- collisions of boats with reef formations;
- hydrocarbon pollution from oil and gasoline from the boat engines;
- pollution due to sunscreen oil from bathers;
- illegal fishing; and
- discharge of sewage into the Nichupté Lagoon, which leads to elevated nutrient levels in the water. The diversity of marine species at Punta Nizuc has declined, and nutrient levels from pollution by organic waste have increased.

To limit physical damage to the reef, the small boats are required to moor at any of 10 fixed-point moorings that have been established to reduce damage from laying anchors on the reef. These fixed moorings are truly crucial for the survival of the corals. They need to be maintained, and enforcement of the no-anchoring rule may be necessary. The large boats dock at a small artificial island that serves the same protective function while offering services (restaurant and sunbathing) to the tourists.

Under the Ministry of the Environment, Natural Resources, and Fishing, park management has undertaken several activities to manage the use of the marine park. In addition to the moorings, the park management requires a minimum of one guide per 10 people in the water and requires that each guide take a one- to two-day course in reef ecology and care. All visitors are required to wear life jackets, which limits contact with the reef by preventing people from diving. The park also sells biodegradable sunscreens. In addition, the management requires that all boats be licensed by the National Institute of Ecology. However, monitoring is difficult and appears insufficient: of an estimated 1,914 two-person motorboats in the park in 1997, only 638 had permits, even though no fee was then charged for obtaining a permit.

Park Management and Funding

A combination of zoning and small technological modifications has the potential to protect pristine reefs for posterity, research, ecosystem functions, and advanced ecotourism diving while maintaining the “sacrificial reef” in a sufficiently viable condition to satisfy the average tourist. All that is needed is management—which may turn out to be the most difficult part.

NMP management has long faced a severe shortage of staff and resources to manage tourism activities and monitor ecological conditions. The federal government pays only for a core staff of five people, and federal budget constraints make any additional funding unlikely. An additional part-time staff of about a dozen is paid by the nonprofit Commission for the Protection of the Reefs, established in 1995. This commission initially raised all its revenues from voluntary donations. Boat operators had pledged to contribute 150,000 pesos per month (US\$1 $\approx$ 11.7 pesos), but less than half has been collected. The park’s budget was thus based on highly uncertain revenues and barely covered the salaries of its personnel, leaving almost nothing for equipment, fuel, and maintenance costs.

In 2001, an evaluation by John Dixon at the World Bank and Mexican colleagues from the National Marine Park and the ministry in Mexico City suggested the implementation of a park fee. Their study indicated that even a modest fee would dramatically change the budget situation: a US\$1 fee per visitor would increase revenues by more than 300% (under the reasonable assumptions of 90% collection and that the fee would have no effect on demand)⁸. When the park management suggested charging such a fee for each

visitor taken into the park, the boat operators strongly resisted it. Their main objection appeared to be that they did not want to provide information that would reveal their true level of income and thus lead to higher income taxes. However, in spite of their initial opposition, the fee was finally implemented in 2010: “The Federal Marine Park Tax” ranges between US\$2 and US\$5 depending upon the destination, and it is intended to be used only for protecting the waters and the coral reef areas.

Note that in this case, there was no demand-side problem in the sense that the fee could easily be passed on to the tourists. The reason for a rate increase would be not to reduce demand but to secure funding for a public good (or club good) kind of regulatory agency. On the other hand, one could argue that the park charges should go to the central treasury and that the government should pay properly for park rangers. The risks with local fees include suboptimality of various kinds. Among others, it may be easy to fund parks such as the NMP, which have many visitors, but difficult to finance bioreserves with fewer tourists. However, any pragmatic solution is much better than no solution. If fees can be set so that the park’s revenues are roughly sufficient to pay for good management, then a decentralized solution that avoids the transaction costs of dealing with central bureaucracies may have more advantages than disadvantages.

Payments for Ecosystem Services in Costa Rica

Historically, it has been the responsibility of governments to ensure biodiversity protection and provision of ecosystem services. As discussed above, some of the instruments used to achieve such objectives have been the management of resources by government agencies through national parks and the development of programs to encourage private management. Nevertheless, conservation of ecosystems requires financing on a scale many times larger than is feasible from public and philanthropic sources. It is essential to find new mechanisms by which resource owners and managers can realize the economic value created by good stewardship of ecosystems. In recent years, several factors have stimulated the development of markets for ecosystem services. Through these markets, consumers, producers, investors, and donors make payments to individuals or groups that protect ecosystems and thereby supply public services of global ecological value. Since many valuable and biologically diverse ecosystems are located in low-income countries, such payments have global appeal while lessening the conflicts between conservation and local welfare.

Currently, more than 300 payments for environmental services (PES) programs are being implemented worldwide (Blackman and Woodward 2009). In the developing world, Costa Rica has led efforts to experiment with the application of these mechanisms. Environmental services have been

BOX 28-1 Bioprospecting Contracts in Costa Rica

The Costa Rican National Institute of Biodiversity (INBio), a nonprofit research organization created by a group of Costa Rican and foreign scientists, was initially founded to conserve and catalogue the country's genetic resources. In 1991 INBio signed a highly publicized bioprospecting agreement with Merck, Sharpe, and Dohme, Inc., a major U.S. pharmaceutical company. The INBio-Merck agreement called for INBio to provide Merck with plant, insect, and microbiological samples that could be tested for biological activity. The arrangement included up-front payments to INBio for samples, training opportunities for Costa Rican scientists at Merck facilities, collaborative projects with Costa Rican universities, transfer of processing technologies to INBio, and royalty arrangements developed in the event that marketable pharmaceuticals resulted from the collaboration. For its part, INBio undertook the collection, taxonomy, and processing of samples to Merck's specifications.

INBio's experience is probably the first successful case in the context of a developing country where a significant financial return to biodiversity has been implemented through bioprospecting contracts. By the end of 2000, INBio had signed another 11 bioprospecting agreements (Camacho et al. 2000), setting an important precedent for countries interested in financing biodiversity conservation.

recognized for a long time in Costa Rica (see Box 28-1), although clearly the attention they have received, both locally and internationally, has increased in the last decades. By 1999, the country's system of national parks, biological reserves, and other protected areas covered 24.8% of the national territory, and tourism had surpassed coffee and bananas as the country's prime source of foreign exchange. A 1996 forestry law created a financial mechanism and institutional structure to compensate private forest owners for the services provided by forest as a land use through the PES program.

The National Forestry Financing Fund (FONAFIFO), the administrator of the PES (along with a series of governmental organizations), coordinates and prioritizes which areas are to receive payments. FONAFIFO's funding comes mainly from a tax on fuel, an extra tariff on utility bills, international donors (such as the Global Environment Facility), and private or public companies.

The law recognizes four services provided by forests: watershed protection, scenic beauty, carbon fixation/sequestration, and biodiversity conservation. The PES program pays forest owners for producing this bundle of four environmental services through three main activities: reforestation, forest management for timber production, and forest preservation. An annual per hectare payment is paid as compensation to landholders for the bundle of all four services, and no distinction is made between ecosystems or any of the individual environmental services, but a larger payment is made in the case of reforestation. To participate in the program, a minimum of 1 ha is required for plantations while the minimum for forest conservation is 2 ha. The maximum area that can receive PES is 300 ha, except for Indian reservations, which can have a maximum of 600 ha (Rojas and Aylward 2003).

In all cases the payments are made over a five-year period according to a fixed schedule. However, the landowner commits to manage forestry plantations for a period of 20 years (five in the case of forest conservation). This obligation is noted in the public land register and applies to future purchasers of the land.

Although this program seems promising, unfortunately it has had little effect preventing deforestation so far, since it has mainly enrolled lands which would have remained forested even without such payments. The voluntary nature of the PES program and variation across landowners in relative profits from clearing have led PES enrollment to be dominated by parcels so poor for agriculture that they would never be cleared in any case (see Pfaff et al. 2008). To improve PES performance, FONAFIFO will have to make some changes to increase focus on more at-risk parcels and payments to areas under higher threat of clearing due to their higher returns. Some of these changes have been implemented already; targeting does appear to be increasing within the PES, but adjustments in the payments seem more difficult to implement since they require the agency to have information about rents.

Shaping Ecosystem Policy

The case studies selected in this chapter concern preservation in one form or another. Understandably, not all ecosystems can be preserved since it would be prohibitively expensive⁹; some areas will be developed for commercial resource development. This tendency points to the importance of zoning as an overriding policy instrument. Ecosystems and the services they provide (see Chapter 22) must be sufficiently well understood to be able to determine the size of reserves and which other conditions (e.g., buffer zones or connecting corridors) are required to protect ecosystem functions and biodiversity in a satisfactory manner.

Zoning is thus necessary but in itself insufficient, because protected zones will not necessarily be respected if the appropriate protection and institutions are not provided. To achieve effective protection and ensure the due process and legitimacy of the decisionmaking, policymakers must consider the interests of the stakeholders, asking, for example, how they would have used an area in the absence of protection. Protection may be accomplished through a CPR management approach, for instance, or through hybrids between CPR and more centralized management, like in CAMPFIRE.

CPR management is best suited to situations where local resource users have evolved cooperative patterns of resource use or at least can be expected to do so. It is not appropriate for protecting the high seas, Antarctica, or outer space. Nor would it make sense in some areas of tourism; where particular skills are required, specialized public or private agents may be best at providing protection. This does not mean that the public should not have the right to share in the benefits. They should, but at the same time, whatever organization operates a park or other area will also need to recover its costs and make

a reasonable profit. For private and public bodies alike, financing and cost recovery are vital issues that must be faced.

An ecosystem may contain many resources—water, fish, forests, and wildlife. All the policy instruments that can be used to manage these resources individually may also be considered for marine environments and other ecosystems: taxes, subsidies, permits, appropriate tariffs, regulations, and property rights. However, because of the inherent complexity of ecosystem management, the design of instruments will also be more complicated. Interesting instruments for ecosystem protection include zoning, stakeholder participation, and common property rights (and the definition of property rights in general). Certification and labeling are other interesting instruments that have been used in fisheries, farming, and forestry to signal some form of management with an ecosystem perspective or care in the commercial handling of natural resources.

Finally, as mentioned in Chapter 26 and here, many ecologically interesting sites are being protected thanks to payments from governments, private users, and other downstream organizations. Markets for ecosystem services are expected to grow rapidly in the next decades, especially those related to carbon sequestration (see Chapter 27). Connecting global and national action on climate change to biodiversity conservation might help to enhance and protect livelihoods. Although currently the Clean Development Mechanism (CDM) can be used to finance forest restoration, the scale of forest carbon under the CDM is too small to have a major impact (Jenkins et al. 2004). It is imperative to structure the climate change mitigation and adaptation responses in such a way that they are also designed to protect and restore biodiversity and ecosystems. Voluntary markets for carbon might also be shaped to pursue biodiversity goals. New institutions to provide the business services required in ecosystem service markets will be needed in that case. These institutions must encourage transactions by reducing transaction costs and providing better methods of measuring and assessing biodiversity.

Supplemental Reading

Alpizar et al. 2008a, 2008b
Brock and Xepapadeas 2003
Brown 2000
Bulte et al. 2008
Dasgupta 1982

Dixon and Sherman 1990
Eicher and Tschirhart 2007
Polasky and Segerson 2009
Vardas and Xepapadeas 2010

Notes

- 1 Such failures are an obvious and natural feature of experimentation with cultivation methods and institutional arrangements (Ostrom et al. 1999 is a follow-up

to Hardin's famous "Tragedy of the Commons" article). Some failures are a necessary aspect of learning. They become more problematic in the global commons because experiments at that scale leave little room for failure. At the local level, however, failure may be part of an optimal learning strategy if the number of substitute sites is sufficient.

- 2 This section has benefited from comments and information from Edwin Muchapondwa.
- 3 Zimbabwean elephants were estimated at fewer than 4,000 a hundred years ago but now number almost 70,000. They eat 150–300 kg of vegetation and drink a few hundred liters of water each day. In some districts, such as Sengwa, about 75% of the tree cover was destroyed by elephants between 1960 and 1975 (Child, Ward, and Tavengwa 1997). Between 1991 and 1996, 368 people were killed in Zimbabwe in connection with protecting their crops or houses from elephants.
- 4 For more information, visit www.efdinitiative.org/themes/conservation.
- 5 This section builds on Andersson and Sterner 1998.
- 6 More recently, multimillionaire (and "deep" ecologist) Douglas Tompkins became the largest private park owner in the world. He owns 21% of the Palena province in southern Chile (323,646 hectares after purchases in December 1999). In 1997, in response to criticism from the Chilean government and the private sector, Tompkins said that he would turn over the land to a local nongovernmental organization, *Fundación Pumalín*. A striking example of a transaction that is both private and cooperative is the sale of a part of the Easter Island National Park to the islanders, many of whom consider the sale illegal (Goldmann 2002).
- 7 We are grateful to John Dixon for material and comments on this case study.
- 8 The park administration estimated more than 550,000 visitors at Punta Nizuc in 1997, up to almost 800,000 a year by 2009. Boat operators charged US\$38 for a "jungle tour" of the mangroves and US\$35 for a direct trip. If the tours were booked through a hotel, the hotel kept a commission of 10%. Thus, a tax of US\$2 to US\$ 5 would not be a big addition. Estimates of divers' consumer surplus were generally much higher—for instance, US\$15 (Davis and Tisdell 1995, 1996) or more (Wright 1994).
- 9 For example, it has been estimated that \$1.3 billion would be required to fully compensate inhabitants in just nine main African parks (Cernea and Schmidt-Soltau 2006).

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PART VII

Conclusion

The last part of this book is dedicated to synthesizing the effects of the many complexities in designing environmental and natural resource management policy. This task is far from trivial. Real policymaking must address: distributional concerns; uncertainty; information asymmetries; market power; technological progress; and many political, cultural, and psychological dimensions—not only ecological and technical complexities. Furthermore, in many developing countries the lack of resources to undertake environmental protection represents an additional constraint. The challenge is to develop policy instruments that solve the issue at hand while minimizing political and distributional concerns and still promoting efficiency and long-term sustainability.

The problems ahead are significant—especially when other concerns such as economic growth and poverty alleviation are entrenched at the top of the priority list of many short-sighted leaders. Environmental protection and poverty alleviation are not opposite priorities. Many environmental problems of concern to vulnerable groups in marginal areas, generally quite localized in nature, revolve around issues such as the availability of clean water, land degradation or soil erosion on farmland, or the progressive shortening of fallow periods. These affect the poor because they are directly related to household food security. They are also directly related to economic growth

as the degradation of the resource base generally translates into declines in production or income.

The problems ahead require greater focus on protecting the environment. Much has been done during the last decades and much can be learned from the many policy experiences described in this book. Environmental degradation is the greatest challenge of the twenty-first century. The time to take action is now.

Policy Issues and Potential Solutions

The world faces serious environmental and resource problems, but new technology offers numerous ways of solving them. Too often, however, potential solutions are not used because the rules that govern modern economies do not provide strong enough incentives. In some cases, the technology is not yet available, and sophisticated socioeconomic structures are needed to create the right incentives for research.

Environmental policymaking is not a simple choice between command-and-control instruments and market-based instruments (MBIs). Fortunately, the range of choices is richer—as in general economic policy, which is not limited to a choice between planning and laissez-faire. To function well, society needs intermediate policies with a lot of fine-tuning. And to meet several goals at once (e.g., efficiency, sustainability, and fair distribution), a combination of policy instruments usually is required.

An array of policy instruments has been designed specifically for environmental and natural resource issues. In addition, most other policies are highly relevant for resource management, ranging from the definition and enforcement of property rights, to the efficiency of the court system, to macroeconomic variables such as the rate of interest and the exchange rate.

Market-based policy instruments can be designed in many ways, of which Pigovian taxes and tradable permits are just two archetypes. Taxes, charges, deposit–refund systems, and other two-part instruments have been used in northern Europe, the formerly planned economies of Poland and the Baltic Republic of Estonia, and in some developing countries (China, Malaysia, and Colombia). Tradable permit schemes, used for pollution abatement in the United States and in some other countries, are particularly successful in fisheries management. Information provision, labeling, liability, and many other schemes broaden the menu of policies being used. Sometimes policies such as energy taxes and subsidies are “inadvertent” environmental instruments; they were not formulated with primarily environmental goals in mind.

Despite economists’ recommendations to have one instrument for each goal, actual policies, such as energy taxes, are shaped out of complex bargaining processes and reflect a multitude of sometimes contradictory goals. They

therefore typically lack the purity of textbook environmental policies, but so do policies aimed at furthering democracy, participation, equity, and other goals. Energy taxes still provide a good illustration of the way environmental charges work. In other areas, physical licensing and command-and-control instruments are predominant. This should not upset economists because, according to economic theory, physical regulations are appropriate in many cases. In some cases, economic theory may even lead one to recommend the prohibition of certain processes or substances as the most appropriate instrument.

Policymaking Criteria

Economists generally assume that the overriding criterion for society is welfare maximization and that this welfare can be measured as a function of individual utilities. However, the utility and welfare functions may be too complicated to be operational, and it is common to have several separate subgoals. The most prominent are cost-effectiveness, efficiency, incentive compatibility, distributional and equity concerns, and administrative feasibility and flexibility. These goals are neither perfectly clear nor completely separable, and the political process is often a struggle in which groups place different emphases on different goals and have different interpretations of them.

For our purposes, cost-effectiveness means that the instrument achieves the environmental goals at least cost. Efficiency is a more ambitious concept that includes the optimality of the goal (i.e., level of abatement or resource stock) itself. Incentive compatibility means that the agents involved (particularly the polluters but also regulators, victims, and others) have an incentive to provide information and undertake abatement as intended. By distributional and equity concerns we mean that the distribution of costs should be perceived as “fair.” Administrative feasibility means that the instrument is practicable—its operation should not incur excessive monetary or informational costs.

These criteria interact. For example, polluters who think a particular distribution of costs is grossly unfair typically will resist and try to stop implementation. They will not have an incentive to collaborate, and this situation will ultimately lead to inefficiency, particularly where there are asymmetries of information or power. Policy formulation and implementation have many political, cultural, and psychological dimensions. It is therefore important to respect and follow the traditional rules for decisionmaking—sometimes referred to as due process—without naively opening up opportunities for corruptive lobbying. Given the sometimes rapid changes in technology, ecology, and of the current understanding of them, any instrument used must be flexible enough to allow for adaptation to new circumstances.

The criteria vary in importance depending on the conditions that characterize the issue at hand. In an economy with an even distribution of income and when regulating environmental problems with moderate abatement

costs, equity issues are less important. Conversely, when dealing with major issues that affect health (and ultimately life) in countries with large income disparities, distributional concerns may be perceived as equally important as (or even more important than) cost-effectiveness in a narrow sense. When dealing with markets characterized by powerful monopolies or serious information asymmetries, the issues of incentive compatibility may dominate. With complex ecosystem interactions, flexibility is often an overriding concern in determining the design of the appropriate instrument. The policy selection matrix can give some structure to the choice and design of policy instruments. The matrix can be used to compare policy instruments according to different criteria depending on the ecological and economic features of a particular issue. Among the ecological features or conditions that should be taken into account are threshold effects and synergies, for instance. Among the economic features are market conditions, such as the degree of competition, the occurrence of missing markets, and the assignment of ownership rights.

In the rest of this chapter, we present some of the salient issues related to efficiency, risk, information, ecosystem complexities, the provision of public goods, feasibility, market structure, general equilibrium effects, cost distribution, and the politics of national and international policymaking.

Efficiency

Efficiency is the most classic economic argument, often used in favor of MBIs. In some cases, however, other instruments may be more efficient. Care needs to be taken with respect to the type of efficiency intended (e.g., static or dynamic) as well as other conditions, such as the characteristics of cost and damage curves.

Efficiency with Heterogeneous Abatement Costs

When pollution is mixed but abatement costs vary strongly, cost savings can be considerable if the firms with the lowest costs do most of the required cleanup. This fact provides a strong argument in favor of MBIs. Taxes and tradable permits are particularly applicable, then deposit–refund schemes, and, in principle, subsidies (although they have other disadvantages). These instruments save costs by encouraging gains from trade or “specialization,” which is one of the strongest advantages of the market mechanism.

Good examples of specialization are found in the areas of energy economics and global climate change. One reason is often simply the differences in a facility’s vintage—and thus the varying number of years of useful life left—or the years left until the next major refurbishing. The marginal cost of decreasing the atmospheric concentrations of nitrogen oxides (NO_x) or sulfur oxides (SO_x), precursors to acid rain, or of greenhouse gases varies

enormously across the transportation, industry, forestry, and agriculture sectors. It also varies within each of these sectors and between countries. Several European countries have a good deal of experience with energy taxes, whereas the United States has experience with permit schemes. Many interesting examples of MBIs are found in developing or transitional economies.

Efficiency with Heterogeneous Damage Costs

Sometimes it is the cost of environmental damage, not abatement, that varies strongly (e.g., health costs from vehicle emissions between densely populated and less inhabited areas, or the nuisance of noise between night and day). In these cases, efficiency dictates a corresponding variation in the stringency of abatement efforts. Such a variation is in principle possible with various policy instruments. However, price instruments may be difficult to differentiate (particularly when product taxes are used as proxies for emissions fees), making quantitative rules such as zoning or time-dependent regulations more effective. The environmental (health) damage caused by one mile driven by an average car will be hundreds or thousands of times higher in a city center than in the countryside, yet the price of gasoline can vary only marginally.

Some solutions combine physical requirements (e.g., “clean” cars in some zones) with the use of differentiated road pricing. The banning of polluting vehicles with two-stroke engines is a much-needed zoning policy in some of the world’s most polluted cities.

Intertemporal Efficiency with Technological Change

The cost of abatement technology may change drastically over time. If costs decrease due to technological progress, for instance, then the effect of a tax, permit, or standard will change in different ways. Progress in abatement technology means that achieving abatement becomes cheaper and that the optimal level of abatement increases while the necessary tax level decreases.

With technical progress in abatement, (constant) taxes produce more-than-optimal abatement, whereas (constant) regulation gives less-than-optimal abatement. The size of the respective losses depends on the relative slopes of the damage and abatement cost curves. However, time commonly increases estimates of the damage from emissions through various mechanisms, including increased population, income, and knowledge. In this case, the tax (with its higher abatement level) is preferable. On the other hand, it may be easier to set physical standards, which can be based on observed technological prototypes, whereas the corresponding tax levels must be estimated. The ability of the instrument to give efficient or at least reasonable outcomes in the face of changing conditions (ecological or economic, such as growth or inflation) is a form of flexibility that can be very important.

Intertemporal Efficiency with Inflation

The effect of inflation on policy is similar to that of technological change. However, one difference is that both abatement and damage costs are changed equally, and thus the optimal abatement or emissions level remains constant while the necessary tax level increases. Consequently, regulation and standards have an advantage for preventing environmental degradation in the face of inflation, whereas monetary instruments such as taxes will be reduced in value. Indexing can be a solution in principle but also entails various problems. In the case of subsidies, which have some undesirable long-run effects, it might actually be an advantage if inflation reduces their real value.

Incentives for Innovation under Technical Change

Building good incentives for innovation is a crucial feature of policy design when the potential for technical progress is significant. Such incentives are almost completely absent if the policymaker chooses design standards. With emissions or performance standards, they are somewhat greater, but the instruments with the clearest incentives are the MBIs and particularly taxes. This issue is nicely illustrated by the effects of the Norwegian carbon tax, which has led to sequestering and underground storage of carbon in saline aquifers.

Uncertainty, Risk, and Information Asymmetry

Efficiency is not always enough or even appropriate. In a world of uncertainty and risk, other criteria—such as producing feasible, verifiable, or robust results—may be more important.

Monitoring Difficulties

Emissions are often difficult to monitor. The consumers of fossil fuels (e.g., gasoline and diesel), for example, are many, small, dispersed, and mobile polluters. In instances where there is a strong one-to-one relationship between the consumption or production of a product and the emissions it produces, then input, output, or product taxes may be good proxy instruments. Taxes on fossil fuels are a good example; they work well to address global climate change, which depends heavily on total (fossil) fuel consumption. However, gasoline and diesel taxes are not very good at internalizing local environmental damage. The emissions from a car and the health impact of those emissions increase with the amount of fuel used, but they vary more significantly with other factors, such as motor and emissions technology. When monitoring

becomes sufficiently difficult, the regulator may choose instruments that specifically facilitate monitoring, such as design standards.

Ex Post Verification

Inappropriate disposal of hazardous wastes is an externality that policymakers must try to reduce. The trouble is that the amounts of effluents and emissions are so small, irregular, and dispersed that they are almost impossible to monitor. It is not the production or use of the product but its disposal that is environmentally damaging, and therefore, product taxes are hardly appropriate. Deposit–refund systems may be the ideal mechanism. They are effectively a tax on disposal but, for the sake of convenience, are levied at the time and point of purchase. The burden of proof concerning appropriate disposal is placed on the buyer, who can avoid paying the “emissions tax” by turning in the item to collect a refund. A similar concept is the “presumptive” tax, whereby consumers are taxed because they are presumed to pollute unless they prove otherwise.

Deposit–refund systems are widely used for glass bottles and aluminum cans. For environmental reasons, they are also used for cars and batteries in some countries. Similar mechanisms could presumably be extended to a wider range of hazardous products. The general principle of the deposit–refund scheme could be used to allow vehicles that can prove they are cleaner to get refunds on annual registration or other taxes, for instance.

Monitoring Only of Ambient Conditions

A common situation that contains elements of asymmetric information is nonpoint-source pollution when only aggregate (not individual) emissions can be monitored—agricultural runoff of nutrients that lead to eutrophication for example. Information is asymmetric because each party to be regulated (i.e., each farm) has at least some notion of its own emissions, whereas regulators can at best monitor only aggregates, such as the ambient concentrations at some points. In this case, emissions taxes are not feasible.

In principle, regulators might levy some form of an ambient tax based on total ambient levels, but this instrument has considerable problems of practicability. In some cases, input taxes on fertilizers or pesticides might be appropriate, but not all agricultural runoff is caused by purchased inputs. It may result from animal manure or soil erosion, which are difficult to monitor. One possible policy in this case is peer monitoring: neighbors typically know about each others’ production methods. If cooperation and reciprocal monitoring can be organized, then this approach may be much more efficient than outside monitoring. It is in fact an essential comparative advantage of common property resource (CPR) regimes. One policy instrument may be to encourage the operation (and perhaps even the formation) of such CPR systems.

Even when no CPR management system is in place, collaboration between neighbors might be elicited; regulators might give a group of polluters a choice between self-regulation and some other policy (e.g., input control or tax). Similar policies may also be applied to the “informal” industrial sector, as is done with two-tier monitoring in Indian industrial estates.

Uncertainty When Damage Costs Are Steep

With perfect certainty, there is a symmetry between price-type and quantity-type instruments. A tax of T dollars per ton of effluent will essentially have the same effect as requiring a mandatory emissions limit equal to $\hat{e}$ tons if the marginal cost of abatement is T at the level $\hat{e}$. When there is uncertainty about the abatement costs, the symmetry breaks down; if the marginal damage costs are steep, then quantitative permits should be preferred over taxes or fees (e.g., when there are dramatic threshold values, such as for many banned pesticides, including dieldrin). Policymakers cannot incur the risk of searching for the optimal tax level; they must simply ensure that there is a safe standard. The costs of a small mistake in the quantity used can be terribly high, and the precision of the quantitative instrument in this case is higher.

Uncertainty When Abatement Costs Are Steep

When the cost of abatement rises steeply and the marginal damage from pollution is relatively flat, an instrument such as a tax is preferable. In this case, the tax has greater precision in avoiding potentially excessive abatement costs. Examples of flat damage curves may be found with stock pollutants (as long as there are no strong threshold effects or nonlinearities). Examples of steep abatement cost curves may be found in the formerly planned economies, where abatement was urgently needed but many companies could not afford new investments. If they were forced (by stringent standards) to switch technologies quickly, it could lead to bankruptcy, unemployment, and significant welfare and political costs.

Uncertainty and Risk

The value of a firm’s formal, legal liability for possible environmental damages is in practice limited by the firm’s ability to pay. If the company goes bankrupt, then its liability may essentially be worthless. Other legal instruments (criminal law), mandatory insurance, or liability bonds—deposits of money that can be used in the event of an accident and that are deposited before a company is allowed to start operations—may decrease this risk. Such instruments are infrequently used because their large costs may be a strong disincentive to business; however, they would be appropriate for those industries that risk really large releases or accidents, such as the Bhopal tragedy.

Missing Markets in Insurance and Banking

Insurance usually is not fully supplied by the market. Information asymmetry causes a market failure through the adverse selection of clients (the high-risk people buy the insurance policies) and moral hazard (once people have a policy, they may become reckless), and these factors lead to an undersupply of insurance. Users of natural resources typically operate at high levels of risk and cannot get the insurance they need. Particularly for the poorest farmers in developing countries, who live close to starvation, the risk of crop failure is unacceptable because they lack insurance, and for similar reasons, they have inadequate access to regular banking and savings institutions. They therefore are very risk averse. The result may be unsustainable behavior: People may not dare to invest in new productive methods and may use methods that damage the ecosystem, such as applying excessive amounts of pesticide or keeping large herds of cattle. These practices, although unsustainable, may be individually rational adaptations to missing markets for savings and insurance. In these cases, the best policy will not be to tax or ban pesticides and large herds but to help provide the missing markets by encouraging insurance companies or microcredit institutions like Grameen Bank.

Ecological and Technical Complexities

Ecological and technical complexity foster uncertainty and thus make the simple efficiency criterion hard to apply. The instruments chosen to deal with that complexity also must consider information, risk, and uncertainty issues.

Complex and Potentially Hazardous Technology

Designing a sufficiently detailed tax for a complex technology (e.g., many chemicals, or large industrial plants) is difficult; a tax must be precise, and it is easiest to levy a tax on a straightforward quantitative measure, such as tons of an input, output, or effluent. Some very sophisticated tax instruments with different tariffs for each chemical and differentiation with respect to factors such as stack height and sensitivity of the ecosystem were designed and used in the former Soviet Union, but the interpretation of taxes in a planned economy is very different from that in a market economy. Passing this kind of tax law in a market economy probably would be difficult. Not only is the level of detail daunting, but regulators explicitly lose control over the quantities emitted. If the tax level is “too low” and a company decides to emit a large amount of some chemical in the middle of a city, then regulators would be in trouble—it would be too late to add extra requirements for abatement. This risk can be thought of in terms of a steep damage curve that favors individual regulation. The relevant criterion is no longer efficiency but rather feasibility and the

precautionary principle of avoiding excessive damage risks. For complex technology, individual licensing or liability may be the best options, particularly if the potential damage is serious (steep marginal damages), as with nuclear and some chemical industries.

When the potential damages and risks are not quite as serious, the best available instruments may be various forms of voluntary agreement or information provision, with or without labeling (e.g., the U.S. Toxics Release Inventory, PROPER in Indonesia, or product labeling). As we saw in the comparison of instruments on the solvent trichloroethylene, fees, prohibition, and restrictions on use may be feasible instruments, but each has its advantages and disadvantages.

Ecological Complexity

Technology is not necessarily the main source of complexity. In many cases, the ecosystem itself is complex, and policies must be guided by the precautionary principle. For biodiversity protection, it would be difficult to construct suitable indicators on which to base taxes. Not only may many features need to be monitored but regulators might want to differentiate among sites as well.

One suitable set of policies is based on the CPR management concept, by which property rights are allocated at the local level and local knowledge of the ecosystem and sustainable harvest methods is harnessed. In fisheries management, individual transferable quotas (ITQs) are a good example of a new policy instrument whereby the issues related to the ecosystem complexity (the total allowable catch) are specifically separated from the issues concerning allocation (of percentage shares) among resource users. In other cases, ecological concerns may favor zoning instruments such as the creation of marine reserves or parks.

The Provision of Environmental Public Goods

In situations that involve some sort of public good (e.g., a national park), no one polluter can be charged or regulated. Available instruments are public provision—and possibly private provision—financed or subsidized by public funds.

CPRs, Local Public Goods, and Mixed Public Goods

Public goods are not always “pure.” Services relating to urban and health infrastructure such as garbage collection are impure (or mixed) public goods or, in some cases, club goods or local public goods. In the past, it was often taken for granted that services such as waste collection should be seen as pure public goods. Lately, discussions concerning efficiency as well as incentives and other factors have led to a modified view. In many high-income countries,

waste management is experimenting with variable charges to provide an incentive for separation at the source, and many low-income countries are experimenting with innovative solutions such as microenterprises to organize workers in this sector. Frequently an inherent conflict arises between the tariff constructions that would maximize efficiency and those that would help provide necessities for the poor.

Ecosystem resources have many “public” characteristics at the local level. The appropriate policy instruments vary with the circumstances. In some cases, communities have started to pay for ecosystem protection of water supplies or other amenities. In other cases, the reform or clarification of private property rights is an important prerequisite for other policies. For true common pool resources (marginal lands, mangroves, fishing sites, and water sources) that are particularly important for the poorest segment of a society, CPR management may be the most appropriate policy.

It is not easy to “create” new CPRs. Social institutions hinge on reputation and social structure, which create trust and reciprocity; such structures take time to build, and the process may be difficult, as illustrated by CAMPFIRE in Zimbabwe.

Global Public Goods

At the other end of the scale are several resources or ecosystem attributes that can be described as global public goods: temperature and climate of the atmosphere or protection against ultraviolet radiation by the stratospheric ozone layer. For global public goods, international cooperation is a *sine qua non* of any successful policy.

The value of local programs may be to demonstrate that abatement or reduced exploitation is feasible. While important, local efforts cannot replace international negotiations or concerted action. The characteristics of the international treaties interact with the choices of policy instruments at the national level. If, for instance, treaties specify quantity targets for each participating country, then quantity-based instruments such as permits will be favored because they best fulfill quantitative goals.

No Polluters Accountable

In some cases, polluters are theoretically identifiable, but monitoring is so difficult that they either cannot be identified or are identified too late (e.g., a company that polluted an area decades ago is now out of business). Society must determine whether cleanup warrants the costs of remediation. If so, public provision of abatement or cleanup and subsidies to firms may be the only methods available to achieve the desired goal, although in some cases, firms, entire industries, and even associated insurers, suppliers, bankers, and customers may be involved through extended concepts of liability.

Feasibility, Market Structure, and General Equilibrium Effects

The social and economic context within which instruments are to be applied is often a more important determinant than the sector or industry concerned.

Number of Polluters and the Structure of Markets

The number of polluters and the structure of markets have profound effects on the choice and design of policy instruments. If there is only one polluter (i.e., a monopoly), then a tax will be passed on to consumers and in fact will have perverse incentives because monopolies are characterized by too low an output level—an effect that may be worsened by a tax. Furthermore, if there is only one polluter, decisionmakers would tend to use individual negotiation, licensing, or voluntary agreements instead of going through the whole process of writing a tax law. (Typically, policymakers also strive to introduce competition and break up monopolies which is an argument against adapting policy instruments specifically to them.)

When the number of polluters is intermediate, the analysis of different instruments can become complex, particularly when abatement is carried out in several steps (e.g., in the acquisition of expensive abatement equipment and the operation of such equipment). Permits are one instrument for which the number of participants is crucial. With few players, the permits will be traded in “thin” markets, which may create significant distortions and in some cases very limited trading. Thin markets also may provide an incentive for strategic behavior by the firms.

Lack of Resources

Countries that are embarking on new or ambitious environmental programs may be constrained by a lack of knowledge and organizational, technical, financial, and human resources. “Sophisticated” instruments might appear to be completely out of their reach, and they might be tempted to conclude that poor agencies should start with command-and-control instruments, leaving supposedly more advanced (market) instruments for later. This approach is unreasonable because all environmental instruments have much common ground; all require systems for monitoring, reporting, verification, and control.

Physical command-and-control instruments are not necessarily easy to administer either. They require a system of penalties and enforcement that must be severe enough to act as a deterrent but not so draconian as to be unenforceable in practice. For this reason, informational, legal, or market-based instruments are sometimes preferred. The sophistication of the instrument may be designed to address an environmental protection agency’s lack of resources. The use of product charges (rather than emissions or effluent

charges) saves on administrative resources but also reduces allocative efficiency. Earmarking these charges can provide resources for an environmental protection agency.

One should also be wary of the simplistic notion that environmental taxes will always provide a win–win situation with abatement and tax revenues. It is not always the case, even though some countries that are well endowed with natural resources probably would benefit from policies designed to capture a greater share of the rent while providing for more sustainable resource use.

Only Subgroups of Polluters Targeted

Sometimes only a subset of polluters is targeted as a result of technical or social factors—possibly because of size or nationality, or because the same pollutant can be emitted from a point source in one kind of industry but a nonpoint-source in other industries. The argument might reasonably be made that it is unfair to tax the identified polluters if other (equally important) polluters are not taxed—particularly if the parties are competitors.

In this situation, policymakers may seek instruments or combinations of instruments that provide an incentive for abatement to those firms that have been identified without penalizing them unfairly with respect to other firms outside the program. This goal might be achieved by presumptive taxation or input taxes that affect the nonidentifiable polluters. Otherwise, refunded emissions payments or the free allocation of tradable permits may provide abatement incentives while limiting the cost burden to the group of firms concerned. Such instruments have been used for abatement of SO_x and NO_x in several countries.

Reducing the Use of Goods that Lead to Pollution

Most analyses of policy instruments are partial analyses, which sometimes are sufficient—for instance, when technical abatement measures or input substitution readily achieve the desired environmental improvement goals. When abatement is technically difficult, environmental goals can be achieved only by reducing the consumption or production of certain goods. The instrument used must have an effect on product output. The general equilibrium repercussions throughout the economy must be taken into account. Simple technology or abatement devices that can reduce carbon emissions sufficiently to slow global warming, for example, are not (yet) available. It will be necessary to reduce the consumption of such goods and services that result in significant emissions of fossil fuel–based carbon (and other greenhouse gases). MBIs such as taxes and traded permits have this output effect and are thus the most appropriate in such cases.

Recycling Revenue from Environmental Taxation

Emissions can be reduced by a carbon tax or by a system of traded permits. Both systems result in an output effect, but only the tax or auctioned permits lead to the public capture of the associated rent. If this rent is captured, then there is a revenue-recycling effect that may be important in achieving overall efficiency. A tax large enough to substantially lower carbon emissions would generate enormous public funds and thereby considerably change the economy. The revenues raised could be used for public spending or for reducing other taxes, and the way they are used could affect the cost of the policy under discussion through secondary effects on the economy and thereby the politics of policy design.

Cost Distribution and the Politics of Policymaking

An analysis of policy instruments should not stop at their general allocative properties or overall resource costs. It is crucial for political feasibility to consider also the distribution of costs. Different instruments result in different cost allocations between polluters and among polluters, victims of pollution, and society. These differences often are decisive in implementation.

Burden of Cost and Issues of Political Feasibility

Technically, congestion and overuse of common goods can be rectified by implementing a tax. However, although the congestion may easily be brought down to an optimal level, the collective welfare of the users may be worse than before the tax. Sometimes users assert historic rights to their level of activity (e.g., fishermen who were “born into” their profession, taking over from generations of fishermen before them), resisting the notion that the government has rights to all the surplus even if it did take fishing down to sustainable levels. In other cases, polluters may be too powerful to tax. In both cases, a tax is politically infeasible, but other policies may be available. Rights can be created—full property rights or at least freely allocated pollution or resource permits. Price-type instruments such as charges also can be used as long as the proceeds are (at least partially) refunded or used in a way that is beneficial to (and determined by) the community concerned.

In the United States, road charges are earmarked for road funds, and in many developing or transitional countries (notably, Poland), environmental fees are earmarked for environmental funds. These funds risk the inefficient use of public money, but in some cases, this risk is the “price” that must be paid to achieve sufficient political acceptability. If modifications in instrument design make the use of an instrument feasible (or feasible at a higher level), then this feasibility may warrant the modification. In a choice between a low but pure tax (that goes to the treasury) and a much higher but refunded

emissions payment, the latter may, in many circumstances, be preferable from the perspective of both environmental concern and overall welfare.

Rent-Seeking and Political Economy

Public policies are not only formed by abstract considerations of optimality but through lobbying and the interplay of various interest groups. Policymakers should anticipate this behavior and be particularly cautious about instruments that tend to promote it. The most obvious example is subsidies, which not only are expensive but also can promote lobbying and even corruption. Even the allocation of permits and the mechanisms for refunding charges may attract considerable lobbying, and these consequences must be taken into account.

These issues are likely to be important if the targeted polluters:

- are regionally or ethnically different and feel discriminated against;
- are poor, where welfare considerations become particularly prominent;
- are rich and powerful enough to stop or stall implementation; or
- are a small homogeneous group that can easily organize.

For each type of group, allocation, refunding, and compensation mechanisms may need to be considered separately.

Fairness and Efficiency for Resource Issues Affecting the Poor

The most urgent class of problems is that in which poverty and environmental degradation occur together. The effects of the two problems commonly reinforce each other, so environmental degradation leads to decreased access to water, fodder, firewood, and other important materials. The desperation and short-sightedness caused by poverty may force poor people into unsustainable practices that worsen resource degradation. All the distinct categories of policy that have been discussed are applicable, depending on the details of the individual case. In rural settings, CPR management may be the most appropriate approach.

Difficulties sometimes arise because efficiency requires market solutions, but some market solutions are socially unacceptable. When distributional and efficiency concerns are equally important and problems are complicated by information asymmetry, then various two-part instruments may provide solutions (e.g., increasing block tariffs or more complicated mechanisms sometimes used for water or electricity, including subsidies for necessary consumption by small consumers). Privatization may be beneficial in introducing market principles but is no panacea. If there are distributional, environmental, or other welfare goals in addition to efficiency, then specific public action will be required because an unregulated market will not reach these goals.

National and International Policymaking

With technical progress and minimal attention to environmental issues, simple problems are addressed that relate to the most immediate environment. It is not uncommon that technical fixes do not really “solve” problems but rather move them farther away. Transboundary pollution is becoming more important, as is the management of the global commons. In many contexts, policymaking must be considered in terms of a structure at the local, national (federal), and international levels.

Policymaking in a Small, Open Economy

Trade relations, transboundary pollution, and international treaties are three global layers of restrictions to consider when designing policy instruments. Trade relations are more noticeable in small, open economies than in large, closed economies. In the small, open economy, the price of goods is set by the world market. Any local deviation (e.g., a pollution tax on domestic production) will not change the world market price and therefore will have no effect on consumption. Its only effect will be on profits and thus market shares in the country concerned.

If the pollutant in question is a global one, then imposing an environmental tax risks merely moving production abroad, thus leaving pollution levels constant. In such cases, international negotiation is paramount, but still many other (national) instruments are available: refunded emissions payments, taxes on consumption (rather than production), free permits, voluntary agreements or licensing, labeling, and information provision.

Mixed Global and Local Public Goods or Environmental Benefits

One important category of projects has benefits at various levels. For instance, local benefits of forest protection may include climate stabilization, maintenance of biodiversity, protection of shorelines, moisture retention, and the halting of soil erosion. At the same time, protecting forests has regional and international benefits (such as carbon retention). Of particular relevance are the cases in which local benefits would not be sufficient to achieve conservation but the total benefits might be. By finding mechanisms to compensate the local economies for global benefits, this category of socially profitable projects may become feasible. Debt-for-nature swaps, the Global Environment Facility, and the Carbon Fund are all (fledgling) examples.

Conclusion

One factor that commonly hampers policymaking in many countries (particularly the poorer ones) is the weakness and lack of resources—not only funding but also staff, training, and laboratories and other facilities—in the environmental protection agencies or the ministries in charge of designing policies. The U.S. Environmental Protection Agency, for example, has more than 17,000 employees, not including outside contractors, whereas China—a country with four times the population and significantly more pollution per capita—has only about 300 workers at the Ministry of Environmental Protection in Beijing and perhaps 2600 workers including affiliate agencies and institutes (WRI 2008).

Experience has shown the importance of prioritizing and concentrating on a limited number of issues and only the worst polluters. Institutions that are knowledgeable and free from nepotism and corruption must be built. It also is important to build partnerships with the various stakeholders and for the environmental protection agency to be seen not only as a police officer but also as a source of technology and know-how concerning modern sustainable technology. Information and technology dissemination, research support, and extension services are—in addition to the inevitable control function—essential to policymaking.

The setting of fees is a difficult and sensitive issue. In some cases, low fees that are earmarked for abatement, research, and control may be useful. Creating environmental funds in which the polluters have influence may help build political acceptance among regional or sectoral stakeholders who need to be involved rather than alienated. Effective environmental work often builds on functional partnerships. Clarifying rights—with respect to resources as well as to information and to litigation concerning damages—may be an integral part of achieving successful policy implementation.

In this book, we have tried not to overuse the term “economic” instruments. In fact, the term is not clear, because economic theory commonly suggests the superiority of quantitative, informational, or legal instruments. In some cases, the main option may even be to strengthen moral and ethical rules. (Economic incentives or formal rules might possibly “crowd out” moral imperatives so that they reduce altruistic behavior.) Real-life policymaking is seldom a neutral search for the optimal instrument to maximize global welfare. At least as frequent is the battle between lobbying groups striving for survival, personal benefit, power, or environmental goals. Following transparent, democratic, and bureaucratically feasible processes for decision-making is crucial for successful policy implementation. The parties affected or interested in legislation must be given the opportunity to influence it for the sake of legitimacy, and because they are the best sources of information, the process must be designed so they have an incentive to reveal at least part of this information. On the other hand, the parties cannot be given too great

an influence if it entails the risk of not using effective instruments to attain reasonable goals. To understand the politics of policy design, careful attention must be paid not only to the allocative properties of the instruments but also to the resulting cost distributions between polluters and others in society.

The dominant factors vary strongly from case to case. If abatement costs vary considerably, then efficiency dictates that market mechanisms such as taxes or tradable permits be used. This is the classic argument for the superiority of the market, and if total costs are high, it will be an important factor. If, on the other hand, the damage costs are sufficiently heterogeneous, then more physical, quantitative instruments such as zoning, differentiated regulations, and differentiated licenses may be called for. If important information asymmetries exist, then policy instruments must be designed to be self-revealing, like deposit–refund schemes. If the public good character of the issue is important, then pure research and the dissemination of information will be crucial to successful environmental policy.

In the face of particular technological or ecological complexities, flexibility may be of paramount importance. Policy instruments may be designed to separate decisions concerning the absolute level of resource use from the allocation of rights or the distribution of permits or quotas. In fishing, the separation of these two aspects through ITQs ensures that the catch level can be delegated to expert bodies who can make rapid decisions based on scientific evidence without having to consider or renegotiate all the political complexities surrounding the distribution of costs and rights. This model may be worth considering in other areas, too. In many cases, the power of the polluters is considerably stronger than that of the environmental protection agency. Consequently, instruments such as Pigovian taxes are impossible, whereas tradable permits allow policymakers to fine tune the allocation of rights and cost distributions to make the policy politically acceptable. Similarly, charges that are differentiated and refunded or paid into environmental funds may be used to secure acceptance from important polluters while strengthening the public agencies.

Environmental economics—particularly policy design—is an exciting academic discipline, but it also has a serious purpose. Models are created not (only) to publish in journals but to make a difference in real economies. Sometimes reality requires a great deal of sophistication to match the complexities of technology, ecology, and society. This does not mean that an instrument should be judged only by its complexity; nor should it be judged by how well it fulfills any other single criterion (such as the polluter pays principle). The ultimate test of an instrument is its effect. This effect is determined first by whether the policy gets implemented and second by the level at which the policy is implemented. A perfectly designed Pigovian tax that is set too low may well be much less efficient than another instrument that for some political reason was feasible to set at a higher level.

The resource and environmental problems ahead are significant but probably not insurmountable. It is an important challenge to adapt and develop

the general principles discussed here to strive for a more sustainable economy. This ongoing process must be informed by theory as well as experience. The careful evaluation of new policies and the sharing and comparison of experiences must be integral to this process. It will require collaboration not only between countries but also across disciplines and between different segments of society. Researchers have a very important message to communicate and must learn to convey it in a way that is visible and practical to decisionmakers. Researchers must be humble and practical enough to acknowledge that many policy instruments evolve from the everyday process of regulation and negotiation, rather than directly from textbooks.

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