

Review of Environmental Economics and Policy

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Introduction to the *Review*

If your desk looks like ours, you have a stack of journals you have not had time to examine, let alone read. Added to this are the daily e-mails you receive from a host of online services, universities, think tanks, interest groups, colleagues, and an assortment of others informing you of yet another interesting article, essay, or book. Truth be told, we are all subject to a paper and electronic blizzard, and on many days this information overload becomes a cacophony of competing voices that may best be ignored. Why then is the Association of Environmental and Resource Economists (AERE) launching a new journal? And how can we—its editors—credibly claim that this is a journal you will really want to read?

The *Review of Environmental Economics and Policy* is intended to fill the significant gap that now exists between the popular press and scholarly environmental and resource economics journals. As with many other areas of academic inquiry, environmental and resource economics has become increasingly specialized and increasingly technical. The result is that articles in academic publications—including AERE's highly regarded *Journal of Environmental Economics and Management*—are typically aimed at researchers who share the interests and background of the author. At a minimum, familiarity with specialized jargon and mathematical and technical tools is assumed.

At the other extreme, when academic economists write for the popular press, such as with op-ed or other opinion pieces, their focus is inevitably on the consequences of particular policies, frequently drawing on their political positions. They give considerably less attention to their own recent work (or that of their colleagues) as professional economists; the market for opinion pieces in the popular press rarely provides a welcoming home for lengthy discussions about academic research and its implications.

In order to fill this gap between the popular press and scholarly journals, the *Review* will publish articles that serve one of several diverse goals: to synthesize and integrate lessons learned from active lines of environmental economic research, to provide economic analysis of environmental policy issues, to encourage cross-fertilization of ideas among the various sub fields and perspectives of environmental economics, to offer readers an accessible source for state-of-the-art thinking, to suggest directions for future research, to provide insights and readings for classroom use, and to address issues relating to the environmental economics profession. As we explain below, most articles appearing in the journal will be solicited by the editors, though all are subject to peer review.

If this sounds somewhat like a description of the *Journal of Economic Perspectives*, it is not by coincidence. Our goal is to produce a journal—the *Review of Environmental Economics and Policy*—that is the flagship to the *Journal of Environmental Economics and Management*, as the *JEP* is to the *American Economic Review*. Our ambitious goal is that the

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Review will come to be both an exceptionally well-regarded and well-read periodical within its targeted audience—economists and others in academia, government, the private sector, and the advocacy world who are interested in environmental and resource policy. We are committed to being international in scope and coverage, reflecting the international—and in some cases, global—nature of environmental problems and the international nature of AERE itself. Our exceptional editorial board, which includes leaders in the profession from around the world, reflects this international perspective. In order to maintain the highest standards, the *Review*—which is published by Oxford University Press—will appear only twice per year. And the high quality of writing to which we aspire will be facilitated by the key role played by our managing editor.

Although our aim is somewhat different than the standard academic journal, the *Review of Environmental Economics and Policy* is nevertheless an economics journal with an intended audience that includes not only academic economists but also professional economists outside of academia, as well as policy makers and shapers with an interest and familiarity with the way economics can and does influence environmental policy. The level of writing in the *Review* is designed to reach this broad audience.

Many of the articles that will appear in these pages will have been carefully and sometimes substantially edited, and sometimes rewritten, in order to make them broadly accessible to our targeted audience. This means that rather than focusing on technical and methodological aspects of research, let alone on specific analytical models, the articles in the *Review* will tend to focus instead on the broad lessons that can be learned—for environmental and resource economics or for public policy—from broader lines of research. We cannot guarantee that every reader will find every article in every issue to be of interest, but our aspiration is that every reader will find at least several articles in each and every issue to be of true interest and value.

We anticipate that professors will find many of the articles useful for their classes, particularly for undergraduate courses and for classes in professional schools. Likewise, because articles in the *Review* will frequently explain research findings from a less technical and specialized perspective, it is our hope that this journal will play an important role in keeping economists and others in business, government, and nongovernmental organizations as well as academia up to date on the frontiers of environmental and resource economics. If we are truly successful, the process of forcing our authors (and ourselves) to explain carefully and critically aspects of research that are ordinarily unquestioned and taken for granted will help stimulate new and better research efforts.

Articles published by the *Review* will be commissioned by the editors but also subject to anonymous peer review. Although the submission of unsolicited manuscripts is not encouraged, we welcome proposals for articles in the form of brief outlines or informal queries. All such proposals should be sent to the editorial office, and all will receive prompt attention. One of the reasons for our policy of not accepting unsolicited manuscripts is that we believe it would be a mistake for anyone to write an article for the *Review* without there being a high probability *ex ante* of publication. The journal's relatively unique requirements regarding style and accessibility mean that a manuscript that would be appropriate and acceptable for a conventional economics journal would *probably* not be appropriate for the *Review*. Likewise, a manuscript prepared in the style required for the *Review* would be very unlikely to receive a favorable reception at a conventional economics journal. Thus, our

policy is intended to provide you, the reader, with the best possible set of articles while reducing risk on the part of potential authors.

This inaugural issue is typical of the structure the journal will follow in future issues, with three articles, a symposium, and several regular features. In the lead article, Geoffrey Heal provides “A Celebration of Environmental and Resource Economics,” in which he reflects on the evolution of environmental and resource economics and the relationship of this field of inquiry to economics more broadly. He highlights the fact that some of the very best economists have made fundamental contributions to the development of environmental and resource economics and documents what he describes as a “two-way street” along which methods, insights, and findings travel between our specialized area of inquiry and the larger world of economics research.

Long before other economists—including environmental economists—carried out research on the phenomenon that has come to be called global climate change, William Nordhaus was leading the way with path-breaking contributions. In the second article in this issue, “To Tax or Not to Tax: Alternative Approaches to Slowing Global Warming,” he steps back from the political fray to contrast two fundamental approaches to the problem—quantity-based controls, such as the Kyoto Protocol, and price-based regimes, such as internationally harmonized carbon taxes. His core message is that although the bulk of research and policy discussion has focused on quantitative approaches, price-based methods have major advantages for slowing global climate change.

The real world in which environmental policies must actually be developed, implemented, and evaluated is very different from the abstract world of full information in which analysts can reasonably rely on simple comparisons of the present value of social benefits and social costs. This is the premise behind Robert Pindyck’s survey and synthesis of “Uncertainty in Environmental Economics,” in which he notes that the implications of uncertainty are complicated by the reality that many environmental policy problems involve nonlinear damage functions, important irreversibilities, and long time horizons. The article clarifies the ways in which various types of uncertainties affect optimal policy design, and summarizes what is and is not known about this problem.

This inaugural issue continues with a symposium on what may be one of the most important policy developments of the past decade from the perspective of environmental economics: the European Union’s Emissions Trading Scheme (EU ETS). In the symposium’s lead article, Denny Ellerman, who coordinated the symposium, provides an introduction and overview, coauthored with Barbara Buchner, which focuses on the allocation of the rights to emit carbon dioxide (CO₂) and presents the results from the program’s first year of operation. Their point of departure is the irony that an American institutional innovation—emissions trading—which the United States forced into the negotiations of the Kyoto Protocol and which was resisted by Europe, became, after the U.S. rejection of the protocol, the primary instrument for the European Union’s own compliance with its Kyoto obligations. With coverage of about half the CO₂ emissions originating from a region of the world that accounts for 20 percent of global GDP and 17 percent of the world’s energy-related CO₂ emissions, the EU ETS is by far the largest cap-and-trade system in the world.

The symposium continues with an examination of market and price developments in the EU ETS by Frank Convery and Luke Redmond. To provide background for their

discussion of the emissions-trading market, the article describes the main features of the EU ETS, including its institutional and legal framework. Next the authors discuss the development and actual functioning of the market, including market intermediaries, the size and frequency of trades, factors that have affected prices, and trends in trading volumes and prices. The article also offers some reflections on the likely future of the EU ETS.

The EU ETS is remarkable in a number of ways, including its decentralized character. This is the starting point for Joseph Kruger, Wallace Oates, and William Pizer in the third article in the symposium, in which the focus is on decentralization in the EU ETS and the lessons that can be learned for global policy initiatives. After an examination of the implications of the scheme's decentralized structure, they move on to a broader exploration of how the system can be linked with trading systems in other parts of the world—including the United States—in order to facilitate global trading of CO₂ allowances.

In addition to articles and symposia, several regular features will appear in each issue of the *Review*. Maureen Cropper will edit "Policy Monitor," which will provide brief reviews of policy developments that are of particular interest to environmental and resource economists. The inaugural article in the series, by Frank Lecocq and Philippe Ambrosi, explains the basic workings and tracks the performance to date of the Clean Development Mechanism, the element of the Kyoto Protocol that is intended to involve developing countries in cost-effective control efforts through offsets with sources in industrialized countries. In each issue, Kerry Smith will provide his "Reflections on the Literature." As his first column demonstrates, we anticipate a broad-ranging review of various topics of interest, with special attention given to information sources that environmental and resource economists may not usually scan. Finally, the "Announcements" feature offers timely updates on conferences, workshops, calls for articles, and other relevant news from the world of environmental and resource economics. Future issues of the *Review* may include other features, such as letters to the editor, which will, of course, depend upon the interests and contributions of readers.

The editors hope that the *Review* reflects the interests and responds to the needs of the worldwide membership of AERE as well as the *Review's* broader constituency of readers. The ultimate success of this journal will not be measured simply by citation counts or circulation numbers, but by whether the journal is actually read and hence of real value to its audience. The *Review of Environmental Economics and Policy* is intended to serve your interests, and we—the editors—are prepared to use the journal in innovative ways, limited only by our statement of purpose (and our budget). We welcome any and all suggestions about how the *Review* can best serve your needs.

Robert Stavins, Editor
Carlo Carraro, Coeditor
Charles Kolstad, Coeditor

Welcome

The Association of Environmental and Resource Economists (AERE) is pleased to launch the *Review of Environmental Economics and Policy* with this inaugural issue.

The need for a new journal aimed toward a broad audience has long been recognized by AERE members and has been actively discussed for several years by the AERE Board. The time was finally judged right to initiate a careful review of the need and prospects for such a journal, and in 2003, a publications committee comprised of Kevin Boyle (Virginia Tech), Charlie Kolstad (UC Santa Barbara), Bill Provencher (Wisconsin), and Hilary Sigman (Rutgers) was appointed to undertake this task. They polled AERE members, looked at the situation from every angle, and drafted a detailed prospectus. This prospectus served as the basis for approaching possible publishers and potential editors. AERE acknowledges a heavy debt of gratitude to these committee members for all their hard work in mounting this effort.

Charlie Kolstad took the lead in negotiating with publishers. It became clear that Oxford University Press was seriously interested in publishing this new journal for AERE and was prepared to devote the resources necessary to make the *Review of Environmental Economics and Policy* a first-rate journal. Martin Green at Oxford University Press has been wonderful to work with in the process of turning AERE's vision of the *Review of Environmental Economics and Policy* into reality. Of particular note is that the *Review* is owned by AERE and published as a joint venture with Oxford University.

There are three major motivations behind AERE's decision to launch the *Review of Environmental Economics and Policy*. First, the field of environmental and resource economics has now evolved to the point where substantive advice can be offered on a wide array of issues in a form that is readily accessible to policy makers. As a branch of economics, environmental economics has evolved and matured considerably over the last two decades. The reservoir of both theoretical and empirical knowledge is now both wide and deep enough to be reliably drawn upon to help inform policy debates on a regular basis. The layout and structure of the *Review* is intended to facilitate this.

Second, there is a clear perception that the content of AERE's *Journal of Environmental Economics and Management* has become more rigorous but is not always accessible to a general audience. This is not surprising. There is always a tension between the responsibility to push the frontiers of knowledge for an audience of specialists and the responsibility to broaden the policy relevance of that knowledge, particularly when the information needs to be cast in less technical terms to maximize the breadth of the audience. The American Economic Association recognized these competing responsibilities when it decided to complement the *American Economic Review* by introducing the *Journal of*

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Economic Perspectives, which aims to deliver clear depictions and explanations of major economic questions and controversies in a way that is readily accessible to a broader economic audience. In many ways, the nature, style and readability of the *Journal of Economic Perspectives* has served as a model for the *Review of Environmental Economics and Policy*.

Third, AERE hopes that the *Review* will help broaden the Association's reach, influence, and membership. As a field, environmental and resource economics is a tremendously exciting area in which to work, with strong links to other fields and policy makers; it presents a clear opportunity to help make the world a better place. We hope that the *Review* will help convey this message and will inspire an even broader audience to recognize the relevance and vitality of our field.

As AERE presidents who have observed the development of the *Review of Environmental Economics and Policy* from its inception to the publication of this inaugural issue, we feel like proud grandparents. We could not hope for a better set of editors—Rob Stavins, Carlo Carraro, and Charlie Kolstad—to ensure that the *Review* hits the ground running and soon has a strong impact. True to the public-goods orientation of much of the field of environmental and resource economics, we are confident that the readers of the *Review*—across academia, government, and the private sector—will express their shared interests and pitch in to help the editorial team identify and anticipate the important and emerging needs that will drive the content of this new journal. Rob, Carlo, and Charlie welcome your ideas on how to make this enterprise a success.

Trudy Cameron, Richard Carson, and Anthony Fisher

A Celebration of Environmental and Resource Economics

Geoffrey Heal*

Several years ago, I took advantage of the award of the Association of Environmental and Resource Economists' (AERE) Prize for a Publication of Enduring Quality, of which I am very proud and which occupies a prominent place in my office, to reminisce about the evolution of our field and its relationship to the rest of economics. When the editors invited me to contribute to the new *Review of Environmental Economics and Policy*, I saw this as an opportunity to develop these points further and to use them as the basis for a rather personal and possibly idiosyncratic review of environmental and resource economics. This is what follows: it meanders from the history of our subject to the evolution of the Hotelling Rule and then on via oil markets to climate change, to natural capital, ecosystem services (ESS), greening national income, and sustainability. The choice is, as I said, idiosyncratic, but I hope interesting.

In my talk at the AERE meeting, there were two main points I wanted to convey. One was that some of the very best minds in economics, people of great prominence and intellectual talent, have contributed to building our field. This is something to be proud of. The other was that the street between environmental and resource economics and the rest of economics is a two-way street, with heavy traffic in both directions. Of course, we borrow from the rest of economics, but we also contribute to it as well, perhaps more than we are aware. Again, this should be a source of pride.

These two observations are related. Why do some of the best minds in economics work on environmental problems? In part, at least, because they think that environmental problems raise issues that matter for the whole of economics, and because they think that understanding environmental problems helps us to understand important issues in economics as a whole. Hence the two-way traffic.

Environmental Contributions of Leading Economic Thinkers

Let me try to document these points. The first is the easy one. During the 1970s, when some of the basic theoretical framework of our field was established, many contributions

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2006 is the seventy-fifth anniversary of the publication of Harold Hotelling's article "The Economics of Exhaustible Resources" (1931), which in my opinion marks the foundation of analytical environmental and resource economics. I would like to dedicate my article to a celebration of this anniversary.

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were made by people who went on to receive Nobel prizes. These are people not generally considered to be environmental economists, but who nevertheless felt that environmental problems are sufficiently central to economics to merit the investment of their time. As an example of this point, I have in front of me the 1974 issue of the *Review of Economic Studies*, which was devoted to a Symposium on the Economics of Exhaustible Resources. As the editor, I still have a copy on my shelf. The contributors included Robert Solow on “Intergenerational Equity and Exhaustible Resources,” Vernon L. Smith on “General Equilibrium with Replenishable Natural Resources,” Tjalling C. Koopmans on “Proof for a Case when Discounting Advances the Doomsday,” and Joseph E. Stiglitz on “Growth with Exhaustible Natural Resources: Efficient and Optimal Growth Paths” and also on “Growth with Exhaustible Natural Resources: The Competitive Economy.” All of them, of course, went on to win Nobel prizes. And none are commonly on lists of environmental and resources economists. I doubt that any special issues of journals devoted to labor economics, international economics, or the economics of education had contributions from four people who went on to become Nobel laureates. In fact, there could easily have been more than four such authors in that issue. At that point, Michael Spence was working on environmental issues, with a series of articles on blue whales (Spence 1975), Jim Mirrlees had articles on market versus optimal depletion rates (Kay and Mirrlees 1975), Kenneth Arrow was developing his thoughts on irreversibility and option values (Arrow and Fisher 1974), and James Tobin was working on greening national income (Nordhaus and Tobin 1972).

This profusion of environmental contributions by the leading thinkers in economics was not confined to the 1970s, but goes back further. In many ways, the founder of analytical environmental and resource economics was Harold Hotelling, with his fundamental work on the economics of exhaustible resources, his work on the travel cost method of evaluating national parks, and his very early work applying calculus of variations to the management of fisheries.¹ Hotelling made fundamental contributions all over economics—his model of product differentiation along a line (Hotelling 1929) is still a classic—and he would clearly have been a prime Nobel candidate had there been such prizes when he was alive.

The Two-Way Street

I think I have proved my claim that some very deep thinkers have contributed to building our field. Let me now turn to the other claim that I made above—that there is heavy traffic in both directions in the street that joins environmental and resource economics to the rest of economics. Given the point that I have just made, this more or less has to be true: if really prominent theorists worked in resource and environmental economics, it is because they felt that what they were doing was important for economics as a whole.

I was just speaking of Hotelling, so let me turn first to him. Think for a moment of the famous Hotelling rule² in markets for exhaustible resources, which states that the resource rent will rise at the discount rate. This is an arbitrage-free pricing condition: the return on

¹Hotelling’s collected articles are available in A. C. Darnell (1990). The original reference on exhaustible resources is Hotelling (1931).

²The Hotelling rule states that $(1/p)(dp/dt) = r$, where r is the discount rate and p is the market price of an exhaustible resource. For this simple version of the Hotelling rule to hold, extraction costs must be zero.

the physical asset—the resource—which comes from the appreciation of its price, must equal the return on other forms of capital. With this result, Hotelling was the first to describe dynamic equilibrium in capital markets in terms of the absence of arbitrage possibilities, back in 1931. It took the field of finance almost half a century to catch up and make the Arbitrage Pricing Theory—the idea that the absence of arbitrage possibilities is a powerful way of characterizing dynamic equilibrium in capital markets—a central perspective.

Another interesting example also relates to finance. I mentioned above the Arrow–Fisher work on real option values (as they are now called). This was published in 1974, as was related work by Claude Henry (1974a,b). This was contemporaneous with the publication of the famous Black–Scholes formula for valuing stock options. Black and Scholes (1973), by focusing on a special case, established a very neat and practical formula, which duly took off in terms of applications in finance. But in some respects, the treatment in the environmental literature was both deeper and more general. For example, Pindyck (1991) and others have traced the connections between the environmental and financial approaches to the valuation of options, and the current literature on real option values owes more to the environmental tradition than to the financial one.

There are many other areas of economic theory whose development was driven in part by pressure from the field of resource and environmental economics. Optimal growth theory is a good example. Environmental economists have worried intensely about the “cake-eating problem” because of its relevance to optimal depletion issues and its connection to the original Hotelling article on resource depletion, and they have moved this area ahead. Recall that the cake-eating problem—so called by the optimal growth theorists of the 1960s, and in particular David Gale (1967)—asks how we should divide a cake of finite size between infinitely many generations, and so is at the core of the pure theory of exhaustible resources. More recently, work on understanding sustainability has shed considerable light on analytical issues in the theory of long-run growth. In asking when continued growth is possible in a world of finite resources, environmental economists have increased understanding of the roles of technical progress, factor substitutability and capital accumulation in long-run growth. We know, for example, that even if an exhaustible resource is essential to production, in the sense that output is zero when the input of this resource is zero, there are paths on which output remains positive for all time (Dasgupta and Heal 1979). This is not an obvious point.

Intertemporal welfare economics has also benefited from the work of environmental economists on sustainability. Most of what we know about the relative implications of Rawlsianism, Utilitarianism, and various other axiom sets relating to intertemporal justice and equity comes from the work of environmental economists. In particular, this work has shown that although Rawls’s axioms are appealing and constitute a force for social justice in an atemporal context, their implications are more perverse in the long-run and can justify remaining forever at the status quo. And almost everything we know about valuing public goods comes from environmental economics. This is a field that will come to matter to other areas of economics, though people do not seem to realize this yet.

Recent Applications of the Hotelling Rule

Something that I find intriguing is the recent development of new applications for the exhaustibility paradigm of Hotelling. A true sign of a deep analytical structure is that

it resurfaces in many contexts, and that seems to be happening with Hotelling's model. For example, recent work has noted that underground aquifers are exhaustible. Fossil water—water laid down in aquifers many years ago that is no longer being replenished—is an exhaustible resource, and one whose scarcity rent is growing and could become huge with the progress of climate change (National Research Council 2005). The capacity of the atmosphere to absorb greenhouse gases without radical change to the climate system has also been modeled as an exhaustible resource (Heal 1984). Climate scientists are finding it productive to say that for the rise in global mean temperature to be within 2°C , we cannot emit more than a certain amount of CO_2 (W. Broeker, personal communication). This, in effect, states that the problem of managing climate change is one of managing an exhaustible capacity.

Another related and interesting exhaustible resource is the capacity to store carbon dioxide in underground rock formations. According to some perspectives on climate change, this could be a vitally important—and exhaustible—resource over the next half century (Butt et al. 1999). A Hotelling-type analysis tells us something about how to schedule the use of this capacity over time. It also tells us that the social costs of using it comprise not only the obvious costs of collection and storage of the greenhouse gases, but also a scarcity rent. In the engineering studies that have been conducted to date, these insights into intertemporal scheduling and the full social cost have been omitted, but they make a real difference (Narita and Heal 2006). A very different recent application of this paradigm is to the study of drug resistance: Brown and Laxminarayan (2000) have modeled as an exhaustible resource the extent to which a drug can be used before resistance develops among the pathogens it is intended to kill. The world's stock of biodiversity can be seen as an exhaustible resource, too. Every time a species is driven to extinction, this stock falls in an irreversible way. We are depleting that stock and do not fully understand the consequences.

It is puzzling that in spite of its intellectual power and elegance, a key prediction of the Hotelling model gets little empirical support. The prices of exhaustible resources, net of extraction costs, do not seem to rise at the interest rate.³ There are many possible explanations, perhaps the prime one being that in reality few resources are in fact exhaustible in the sense of the Hotelling model. Most have supplies of varying grades, with the lowest grades available in almost unlimited amounts. This implies a rather different relationship between prices and interest rates (Heal 1976), one that has not been tested empirically.

Obviously, resource and environmental economics is intellectually important: I have noted a few of the significant intellectual developments that originated in the field. It is clearly of practical import, too. Two of the headline grabbers of our times are the price of oil—a classic in the economics of exhaustible resources—and climate change, one of the largest manifestations of external costs and public bads of all time. Hardly a day goes by without either or both on the front pages. On both topics, a little environmental and resource economics goes a long way. Consider the following questions.

³Heal and Barrow (1980, 1981). For a dissenting view that does find support for the Hotelling rule in the case of oil, see Miller and Upton (1985).

Past and Future Oil Prices

What is the difference between the run-up in oil prices of recent years and that of 1979–80? The rise then was actually sharper than now, and to levels that are higher in real terms. Then too there was talk of the end of the oil era, of falling discoveries of new fields, and of the need to move to new technologies. Commentators were predicting oil costs rising to a hundred dollars a barrel (in 1980 dollars!). Oil majors invested billions in “synfuels,” oil derived from unconventional sources such as shale and tar sands. Yet before too long, the price was down in the teens again and all the money invested in synfuels was wasted, written off in a sea of red ink. In all the discussions of today’s high oil prices and what they mean for the future, none of the commentators seems aware of this recent history, and none can explain why things are different today, or why prices collapsed in the 1980s. There is virtually no well-informed discussion of the future of oil prices. Yet basic ideas from our field can bring a little clarity to these issues.

Prices fell in the 1980s because of lags in the responses of demand and supply to prices: shortfalls in 1979 and 1980 sent prices up, and inelastic short-run demand let them stay there for a while. But over a few years, demand did respond through more efficient vehicle choices and other energy-saving measures—as it is doing today, as shown by the precipitate drop in the sales of large SUVs and pickup trucks. And supply responded too—the high prices of 1979 and 1980 created the North Sea, Mexican Gulf, and North Slope oil fields. Without those prices, they would not have come on line when they did. But with high fixed costs and low variable costs, oil fields are really vulnerable to hold-up problems: if the price drops to a level at which entry would not be profitable, production continues nevertheless because sunk costs are, after all, sunk. So, within a few years of high prices, there was a drop in demand and an increase in supply and with low short-run elasticities, the market needed a big price move to clear, downwards this time.

Is the same thing going to happen again? Predicting oil prices is of course risky: lots of experts have made fools of themselves in trying. Retrodicting the 1980s is a lot safer! But even without sticking one’s neck out too far, it is easy to see what issues have to feature in an analysis of the future. Demand for oil is growing fast, and the supply is not. Demand growth could cool but will nonetheless almost certainly continue to outpace the growth of conventional oil deposits. This should push prices up.

But here we need to bring in a concept that loomed large in the exhaustible resources literature of the 1970s—the backstop technology. This is a technology that can replace conventional oil with a large supply of something equivalent, a supply with a horizontal supply curve. If it exists, it pegs the price of oil. At this point, it appears that there may be some backstops available. One is the conversion of coal to oil for a cost in the range of forty to fifty dollars per barrel. Another is the extraction of oil from tar sands—as in the Athabasca Tar Sands in Canada or in the Orinoco Basin in Venezuela—also at a cost of forty dollars. A third is gas-to-oil technology, which converts natural gas into gasoline, again at a cost said to be in the forty-dollar range. This is attractive in places where gas is abundant and where there is no local market. Between them, these three sources could provide a lot of oil. Canada, for example, is talking about matching Saudi output within a decade. So perhaps this abundant supply of a substitute at a price in the range of forty dollars will force the price of oil back down

to that range. But there is a complication: oil from coal, and oil from tar sands, requires massive investment, in the billions.⁴ If there is a chance of the price of oil dropping sharply after the investment is made, rational investors should not invest. So there is scope here for a game in which producers of conventional oil try to scare off investors in backstop products by establishing a credible threat of prices dropping. I am sure that they are thinking in these terms. So far they have not made the threat credible, although the possibility of a price drop is clearly the main factor holding several oil majors back from investing heavily in these backstop technologies. Resource and environmental economists studied exactly these issues back in the 1970s.

This doesn't amount to a forecast of oil prices, but I think it clarifies what the issues are. It doesn't go beyond chapter one of a basic resource economics textbook, but generates insights that are starkly absent from today's discussions.

Climate Change Policies

Can we offer easy insights into climate change policies? Again, the answer is certainly yes, but to a greater degree they are already out there in the debate. The fundamental mechanism of the Kyoto Protocol—cap and trade—is one of our great success stories. Unless you are an environmental economist, it appears to be a fundamentally implausible approach. Recall all the hostility of environmentalists to issuing “rights to pollute” that came with the first large-scale use of cap and trade in the 1990 amendments to the Clean Air Act? And the hostility to the use of carbon offsets in the Kyoto Protocol? It's far from obvious that setting up a market in a new-fangled and rather abstract property right will cure pollution or climate change, so we need to congratulate ourselves on developing the idea and getting it out there into application. Another one of our achievements in this area is just raising the idea that we need to compare the costs of reducing climate change with the costs of allowing it to occur, although this basic insight is still not as widely shared as one might wish. If we could get more of the debate focused on this issue, we would make a lot of progress.

Another simple insight that we seem to have gotten across is that there is a discount rate in this problem and it matters. In other words, some of the intertemporal welfare economics that I alluded to above is now being applied in discussions of climate change policies. Personally, I would like to see more and more intelligent discussions of uncertainty and how that features in climate change policies. There is good literature on this, some of course based on the real option value frameworks that I mentioned before (for a review, see Heal and Kristrom 2002). Although the basic science of climate change is robust, there are uncertainties, particularly in the analysis of the social and economic implications of an altered climate. What will it be like to live in a world where it is on average four or five (or more) degrees hotter? Until recently, most of us probably thought that it would not be very different. I am beginning to change my mind on this; evidence seems to be accumulating that quite small changes in climate—those already in hand—can produce noticeable effects, mostly negative. So I would want to give some probability to the event that a warmer world is really very much less pleasant than the one we have now, though I would not treat this as a certainty. It should be possible to explain this to the general public.

⁴It also requires a vast amount of water, which could prove a constraint on the development of these technologies.

To digress slightly, let me talk more about the hostility to the idea of trading carbon emission permits in the context of the Kyoto Protocol. For the last two and a half years I have been actively involved with the Coalition for Rainforest Nations (CfRN).⁵ This is a group of countries with large tropical forests that are seeking to harness the 1997 Kyoto Protocol to generate incentives for forest conservation. Under Kyoto's Clean Development Mechanism (CDM), there are incentives for reforesting land from which forests have been cleared and for afforesting land that was never forested. The reforestation and afforestation incentives under the CDM work as follows: by growing a new forest and providing a carbon sink, countries generate carbon credits that can be sold to entities subject to the Kyoto Protocol to offset their emissions, so-called Certified Emission Reductions (CERs). But, perversely, there are no similar incentives for conserving existing forests. As deforestation is one of the largest of all sources of greenhouse gas emissions, accounting for about 20% of the total (for comparison U.S. emissions are about 25 percent of the total),⁶ this makes no sense at all. In the original Kyoto Protocol, conserving forests was treated the same way, but in the 2001 Marrakech meeting of the Conference of the Parties, it was ruled out in an amendment. This in spite of the fact that by conserving forests we also conserve biodiversity—forest clearing is the largest driver of species extinctions, giving us a real “two-fer” here: cutting back deforestation can reduce one of the largest sources of carbon emissions as well as conserve threatened species. It can also preserve traditional lifestyles and cultures that are threatened by large-scale tropical logging.

With so much going for it, you would expect that environmental groups would be enthusiastic about using the Kyoto Protocol to promote reductions in deforestation. Yet, in fact, they were the ones who really killed this idea when it first surfaced. Some of them are opposed to it even today. This is all a part of their objection to providing firms in industrial countries with the opportunity to purchase offsets rather than reduce emissions directly. They see this in moralistic terms rather than in terms of economic efficiency: they feel that we “ought” to reduce emissions in high-cost areas, almost in the same way that friars in the Middle Ages felt that penitence was not real unless accompanied by self-flagellation. “No pain, no gain” could be their motto, playing directly into the hands of the interest groups who argue that the costs of reducing emissions are unbearable for the industrial economies. A little bit of environmental economics could go a very long way here!

This opposition notwithstanding, the CfRN is making real progress. The 2005 Montreal Meeting of the Parties to the United Nations Framework Convention on Climate Change (UNFCCC) agreed that deforestation is a major contributor to climate change and set in motion a process for investigating how financial mechanisms can be used within the Kyoto Protocol to provide incentives to reduce deforestation.⁷ The obvious option is to treat deforestation like any other emission and agree that reducing it generates credits. For this to happen, the members of the CfRN would assume positions in the protocol similar in some respects to those of the industrialized countries: they would agree to emissions limits and would then be entitled to credits for remaining within these. The involvement of a large

⁵See www.rainforestcoalition.org.

⁶See the IPCC's Third Assessment Report at www.ipcc.ch.

⁷The motion to this effect was proposed by Papua New Guinea, supported by Costa Rica. I had the honor of being a member of the delegation from Papua New Guinea.

group of developing countries in such a Kyoto mechanism would be a politically salient development.

My close involvement in this process has served to reinforce my belief in something that I always tell my students—a little economics (in this case of the environmental kind) goes a very long way. Working on the research frontiers tends to make us undervalue the most basic principles of our subject, but in fact they are very powerful and represent a way of thinking that adds tremendous value to what otherwise seems to be common sense.

Let me try to summarize my thoughts on what we have contributed to the analysis of climate change. There is a real danger of feeling frustrated that we don't have any single simple clean solution to offer, and of feeling that because of this we have not contributed as much as we should have. But, in fact, we have had a huge and beneficial impact on the terms in which this debate is conducted, and steered politicians and environmental groups away from some very serious mistakes. Our ideas have probably reduced the overall costs of attaining any level of emissions reductions by at least a half and probably much more. Our most basic insight, that the emission of greenhouse gases is an external cost that has to be internalized, is of course the most fundamental one in this area. If we could only get the prices of energy sources to reflect their social costs, we would be a long way towards a solution.

Biodiversity and Ecosystems: A New Paradigm

Another major environmental problem is biodiversity loss. I mentioned it above in the context of climate change and deforestation, and it is true that deforestation links the two because it is a driver of both. But logically the two are separate. I think that as a profession, we have had some difficulty in articulating why biodiversity loss matters, and in finding tools that allow us to address this in a fully satisfactory way. However, in recent years, we have made a lot of progress. I want to quote something that Ed Barbier and I said recently in *The Economists' Voice* (Barbier and Heal 2006):

A new paradigm is emerging in environmental economics. It views the natural environment as a form of capital asset, *natural capital*. This is fully in keeping with what is happening in other areas of economics, where alternative forms of capital are central to analyses that have become influential—human capital, intellectual capital and social capital being notable examples.

Natural capital consists not only of specific natural resources, from energy and minerals to fish and trees, but also of interacting *ecosystems*. Ecosystems comprise the abiotic (nonliving) environment and the biotic (living) groupings of plant and animal species called communities. As with all forms of capital, when these two components of ecosystems interact, they provide a flow of services. Examples of such ESS include water supply and its regulation, climate maintenance, nutrient cycling and enhanced biological productivity.

This newly emerging area of environmental economics is concerned with the identification and analysis and valuation of these ESS. What are they? How do they affect human societies? How do the actions of human societies affect them? In short, what are the *values* arising from ESS and why should humankind care about these values?

In grappling with biodiversity loss and how and why it matters, we are forced to think about ecosystems: biodiversity is a fundamental part of an ecosystem, and its loss is thought by biologists to affect mainly the robustness and productivity of ecosystems. We can think of ecosystems as capital assets. Ecosystems matter economically because they provide services that are of great value to human societies.

This new natural-capital-based paradigm sees the environment as a source of services that improves human wellbeing: these services are ESS and the ecosystems that support them are a part of our stock of natural capital.⁸ The ESS are a return on the natural capital: they are what we get in return for investing in rebuilding or conserving this capital. For example, when New York City invested over \$1 billion in restoring the Catskills watershed, this investment yielded a continuing flow of ESS of great value to the city (see Heal 2000 and National Research Council 2005). If we invest in conserving a wetland, then the wetland is natural capital and the services that it provides, such as water cleansing and wildfowl support, are the services that we get as a return on our investment. According to the Millennium Ecosystem Assessment, “ecosystem services are the benefits people obtain from ecosystems” (Millennium Ecosystem Assessment 2003, p. 53). Such benefits are typically described as follows (Daily 1997, p. 3):

Ecosystem services are the conditions and processes through which natural ecosystems, and the species that make them up, sustain and fulfill human life. . . . In addition to the production of goods, ESS are the actual life-support functions, such as cleansing, recycling, and renewal, and they confer many intangible aesthetic and cultural benefits as well.

This vision of ESS and associated natural capital is important analytically because it is very productive: it allows us to bring to bear on the economics of the living environment a set of techniques from mainstream economics, including welfare economics and capital theory. Prior to the development of this paradigm, it was difficult to bring the analytical techniques that characterize much of modern economics to focus on issues arising from our interactions with the living world. Environmental economics was bifurcating into two separate spheres, “brown” and “green.” The brown sphere contained issues relating to the management of pollution and the regulation of polluting industries, and was rigorous and formal. The green sphere, though no less important, was until recently characterized by a less quantitative approach and an inability to measure the concepts that were key to its operation. I see this as an exciting development because it opens up to analytical economists an area that was previously frustratingly closed. It also provides a natural framework for collaboration between economists and ecologists in understanding how and why biodiversity and its associated ecosystems matter to us.

Because there are no markets for most ESS, the issue of how to value them jumps to the fore. To value natural capital, we need to be able to measure and value the services it provides. In recognition of the importance of these issues and of the absence of widely accepted answers to most of the pertinent questions, the National Academy of Sciences–National Research

⁸See Daily (1997) for an exposition of the ecological perspective here, and Heal (2000) for a summary of the relevant economics.

Council (NAS/NRC) in 2002 set up a Committee on the Valuation of Ecosystem Services, composed of economists, ecologists, and a philosopher: its report was published last year⁹ and provides a review of the techniques available for valuing ESS and their strengths and limitations. Valuing the services it provides is of course a prerequisite to valuing natural capital. One of the main conclusions was that regardless of exactly how one defines and categorizes “ecosystem services,” “the fundamental challenge of valuing ecosystem services lies in providing an explicit description and adequate assessment of the links between the structure and functions of natural systems, the benefits (i.e., goods and services) derived by humanity, and their subsequent values” (see National Research Council 2005, p. 2). Collaboration across disciplines is essential to this task.

Although the NAS report found that, to date, there has been good progress on establishing this “mapping” from ecological function to economic valuation for certain well-defined single ESS of aquatic systems, valuing multiple ESS typically greatly increases the difficulty of evaluation and as a result has yielded fewer successes. In fact, understanding how the biochemical state of an ecosystem affects its ability to provide services and so how human changes to that biochemical state affect service is one of the most important research topics in the conservation field right now. As economists, we can typically describe how economic activity will affect an ecosystem physically, but neither we nor our ecology colleagues have the tools needed to understand how this translates into a change in the services an ecosystem supplies and the value of the natural capital that it represents.

National Income Accounting and Sustainability

Thinking of environmental systems as natural capital, as one of several types of capital that drive a nation's economy along with intellectual, human, social, and built capital, lets us talk in a productive way about sustainability. There are many different interpretations of sustainability, and this article is not the place to review them. However, one interpretation that is certainly attractive is that sustainability involves maintaining intact the value of a nation's total capital stocks. This interpretation of sustainability allows us to deplete some kinds of capital, provided that we compensate by building up others. Maintaining the total value of capital stocks intact is not an arbitrary choice for the benchmark for sustainability: one can show that if a country maintains the value of its capital assets intact, then it can also maintain living standards intact over the long run. The precise statement is as follows: the present discounted value of welfare along an optimal path is increasing, constant, or decreasing as the value of capital stocks is increasing, constant, or decreasing.¹⁰ This result gives us a close connection between capital stocks and long-run living standards and has given rise to a spate of work trying to assess the sustainability of the growth patterns of different countries. The World Bank (2005) has been one of the drivers of this work, with its conclusions to date summarized in its recent book *Where is the Wealth of Nations?* This shows very clearly how the composition of total wealth varies from poor to rich countries, with the former having a larger share of natural capital and the latter larger shares of human

⁹The report is available online at <http://www.nap.edu/books/030909318X/html/>.

¹⁰This is proven in several places. One is Heal and Kristrom (forthcoming). Another is Dasgupta and Mäler (2000).

and intellectual capital. It also shows that for most countries total wealth is constant or rising, though less rapidly than measured national income because of the environmental and resource costs of growth in fast-growing countries such as China. In addition, it shows that for some unfortunate countries, total wealth is falling: their paths are unsustainable and their living standards, in many cases already low, are going to fall.

Take Botswana as an example. It is a relatively prosperous, stable, and fast-growing country in southern Africa. In fact, it is one of the little-known political and economic success stories of the last half-century. Much of its wealth comes from diamond mines. Clearly diamond mining is not sustainable by any definition: diamonds are an exhaustible resource. But Botswana has used its revenues from diamonds to build up other forms of capital and offset some of its resource depletion so that its real total wealth per capita has more than trebled since 1980, in spite of the depletion of its diamond reserves (Lange 2004). Neighboring Namibia, however, while also rich in exhaustible resources as well as renewable ones through its rich ocean fisheries, has been more profligate, and its path is therefore less sustainable. Its per capita real wealth has fallen by about 30 percent over the same period.

Tables 1 and 2 show a more general set of results in this spirit from Arrow et al. (2004). Table 1 shows the results when we compute the rate of change of total capital for a wide range of countries, including two rich industrial countries (the United States and the UK), two rapidly growing developing countries (India and China), one very poor developing country (Bangladesh), one very poor developing region (sub-Saharan Africa), and one oil-exporting region (the Middle East and North Africa). The data cover the period 1970–2001.¹¹ The first numerical column shows domestic net investment, which is the starting point for the calculations and an estimate of investment in physical capital. To this is added expenditure on education, an indicator of investment in human capital. We then add investment (usually disinvestment) in various types of environmental capital. The third numerical column shows an estimate of the social cost of CO₂ emissions, the fourth shows the depletion of energy resources (particularly large for the Middle East and North Africa), and the next is forest depletion, large for Nepal and zero for the United States, where there has actually been regrowth of forests over the period of interest. The final column gives the sum, an estimate of total investment in all forms of capital as a percent of national income. Following the World Bank, we call this genuine investment (GI).

Clearly there are many shortcomings here, and I shall talk about correcting some of them shortly. Among the shortcomings that we do not correct are the inadequacy of educational expenditure as a measure of investment in human capital and the incompleteness of the list of categories of environmental capital whose depletion we include. Both could be serious sources of error, but it has not been possible to obtain data to take this process further. Nonetheless, the numbers that emerge make some intuitive sense. For example, for the Middle East and North Africa, a domestic net investment of +14.72 percent turns into

¹¹To be precise, the coverage is as follows: Bangladesh 1973–2001, India 1970–2001, Nepal 1970–2001, China 1982–2001 (without 1994), Sub-Saharan Africa 1974–1982 and 1986–2001, ME and NA 1976–1989 and 1991–2001, United Kingdom 1971–2001, and United States 1970–2001, and are taken from the World Bank (1997).

Table 1 Derivation of genuine investment as % of GDP

| | Domestic net investment | Education | CO ₂ | Energy | Mineral | Forest | Genuine investment |
|------------------------------|-------------------------|-----------|-----------------|--------|---------|--------|--------------------|
| Bangladesh | 7.89 | 1.53 | 0.25 | 0.61 | 0.00 | 1.41 | 7.14 |
| India | 11.74 | 3.29 | 1.17 | 2.89 | 0.46 | 1.05 | 9.47 |
| Nepal | 14.82 | 2.65 | 0.20 | 0.00 | 0.30 | 3.67 | 13.31 |
| Pakistan | 10.92 | 2.02 | 0.75 | 2.60 | 0.00 | 0.84 | 8.75 |
| China | 30.06 | 1.96 | 2.48 | 6.11 | 0.50 | 0.22 | 22.72 |
| Sub-Saharan Africa | 3.49 | 4.78 | 0.81 | 7.31 | 1.71 | 0.52 | -2.09 |
| Middle East and North Africa | 14.72 | 4.70 | 0.80 | 25.54 | 0.12 | 0.06 | -7.09 |
| UK | 3.70 | 5.21 | 0.32 | 1.20 | 0.00 | 0.00 | 7.38 |
| United States | 5.73 | 5.62 | 0.42 | 1.95 | 0.05 | 0.00 | 8.94 |

Source: Arrow et al. (2004)

Table 2 Genuine investment corrected for total factor productivity growth and population growth

| | Genuine investment | Growth of genuine wealth (GW) | Population growth | Growth of GW per capita | TFP growth | Final growth genuine wealth per capita | Growth GDP per capita |
|------------------------------|--------------------|-------------------------------|-------------------|-------------------------|------------|--|-----------------------|
| Bangladesh | 7.14 | 1.07 | 2.16 | -1.09 | 0.81 | 0.30 | 1.88 |
| India | 9.47 | 1.42 | 1.99 | -0.57 | 0.64 | 0.54 | 2.96 |
| Nepal | 13.31 | 2.00 | 2.24 | -0.24 | 0.51 | 0.63 | 1.86 |
| Pakistan | 8.75 | 1.31 | 2.66 | 2.06 | 1.13 | 0.59 | 2.21 |
| China | 22.72 | 3.41 | 1.35 | -3.05 | 3.64 | 8.33 | 7.77 |
| Sub-Saharan Africa | -2.09 | -0.31 | 2.74 | -3.05 | 0.28 | -2.58 | -0.01 |
| Middle East and North Africa | -7.09 | -1.06 | 2.37 | -3.43 | -0.23 | -3.82 | 0.74 |
| UK | 7.38 | 1.48 | 0.18 | 1.30 | 0.58 | 2.29 | 2.19 |
| United States | 8.94 | 1.79 | 1.07 | 0.72 | 0.02 | 0.75 | 1.99 |

Source: Arrow et al. (2004)

a total savings rate of -7.09 percent after allowing for the depletion of energy resources. These results draw attention to the fact that this part of the world lives unsustainably by depleting an exhaustible resource and is not compensating for this by building up its other capital stocks. Sub-Saharan Africa is also shown to be living unsustainably, a tragic and not surprising result. Allowance for the impact of HIV/AIDS on human capital would probably make its total investment number even worse. The remaining countries all appear from these numbers to have positive total investment and so to be meeting one of the criteria for sustainability, namely that the present value of future welfare obtainable from capital stocks be nondecreasing.

However, all of these numbers omit two factors that could be important: one is population change, a real issue in several countries, and the other is technical change. A higher rate of population growth will presumably increase the level of investment required to maintain constant living standards. This means that the numbers in table 1 will overstate the extent of sustainability with a growing population and vice versa. Technological progress will act in the opposite direction, allowing humans to extract more welfare from a given set of resources. Neither population growth nor technological progress was a part of the theory

that has been developed in this area, and indeed as far as I am aware there is little or no discussion of either of these issues in the literature on sustainability or on optimal growth with environmental resources. Yet intuition suggests that they are important, and the numbers in Arrow et al. (2004) confirm this, indicating a lacuna in the theory developed so far. So we have made two modifications to the data in table 1 to adjust for population growth and technological progress.

Table 2 shows the results of these modifications. The first column is the last column from table 1, our preliminary estimates of genuine savings as a percentage of GDP. The second column gives an estimate of the growth rate of genuine wealth (GW) derived from the previous column using an assumed GDP/wealth ratio, and the fourth gives the growth rate of GW per capita, using the population growth rate given in the third numerical column. This is followed by an estimate of the growth rate of total factor productivity and then the growth rate of per capital GW adjusted for total factor productivity growth. For comparison purposes, the last column gives the conventional figure for growth of GDP per capita. Only two estimates of the growth of total wealth per capita are negative, the same two as before, but many others are probably not significantly positive. The high population growth rates of Bangladesh, Nepal, and sub-Saharan Africa all act to reduce their countries' rates of genuine savings.

Although the methodology differs in some technical details, our results are very consistent with those of the World Bank (2005), although the Bank does not allow for technical progress and covers a much greater range of countries. The Bank concludes that most resource-dependent countries are not replacing the capital that they deplete in extracting their resources and are therefore reducing their long-run welfare potential. In other words, they are living unsustainably.

This type of work, setting out a nation's balance sheet in a way that lets us see the evolution of its assets over time, brings a powerful combination of economic theory and national income accounting to bear on the concept of environmental sustainability and shows that this attractive but elusive concept can be made precise and indeed can be measured.

Market-Based Approaches to Environmental Conservation

Thus far, I have spoken of some of the intellectual contributions we have made and how these have influenced policy debates. But there is another element to our impact that is not associated with specific policy issues but is more general and almost philosophical in nature. This is the introduction of market-based approaches to environmental conservation. I believe that some years ago, Bob Solow described most environmental policies as Stalinist. I can't find the exact reference, so the story may be apocryphal, but nonetheless it's too good not to use. The point is that apocryphal or not, it is absolutely correct: far too many environmental policies have relied on telling people exactly what to do and what not to do. They have been classic command and control policies.

An example that comes immediately to mind is the Endangered Species Act (ESA) (Brown and Shogren 1998). One of the most important pillars of conservation in the United States, the ESA is also arguably one of the most controversial, and this controversy has a lot to do with its command and control nature. The central mechanism through which the ESA operates is the listing of a species as endangered. Once a species is listed as endangered,

government agencies, mainly the Fish and Wildlife Service, have the power to ensure that nothing is done to further threaten the status of that species. In particular, they can prevent changes in land use that have negative effects on the species, which gives them the right to prevent development of the land or to prohibit any use of the land that is prejudicial to the species. While at first sight rational, and representative of how environmental groups typically think about such an issue, this clearly places all the costs of conserving the species on the owner of the land on which it lives, even though he or she may bear little responsibility for the species' predicament. To me, the ESA is a classic of command and control legislation, the antithesis of a market-based approach. Not surprisingly, landowners and the property rights brigade have had this act in their gunsights ever since it was passed, and the current administration is giving them the chance to get some shots on the target.

Just imagine if an enlightened economist had designed this legislation, instead of the environmental lobby. What might it look like? Its central feature might be an incentive to conserve an endangered species. Instead of triggering all kinds of limitations on land use, it might instead trigger payments for the support of the species—so much per animal or per breeding pair per year on your land. The more endangered animals you have on your land, the more the Fish and Wildlife Service pays you. If the payment were high enough, there would be an incentive to encourage and assist the endangered species. Indeed, one can even see landowners lobbying for listing, rather than against, as it would represent an alternative source of income for them. Actually, we have come quite close to this in some cases, cases in which mitigation banking has been applied to endangered species.

Mitigation banking was first introduced in connection with wetlands under the Clean Water Act of 1977 and has now spread to other types of habitat. The essence of mitigation banking is that developers are allowed to use habitat that is threatened and protected, provided that they mitigate by ensuring the conservation in perpetuity of a compensating amount of equivalent habitat elsewhere. The choice of what is a compensating amount and what is equivalent is determined by the appropriate conservation authority. In particular, compensation may be at a rate of more than one for one, which means that more than one acre of land is set aside elsewhere to compensate for the use of one of the original acres. A developer may, in addition, conserve more equivalent land than required and hold this excess to sell to others who want to develop yet do not want to be involved in finding and conserving equivalent space. This process of creating equivalent conserved land in excess of immediate requirements to hold for future sale to others who need to mitigate is mitigation banking and is now a well-developed practice in some areas.¹² Landowners as a group probably neither gain nor lose, but there is a redistribution within the group (Heal 2006).

The classic example of mitigation banking is the red-cockaded woodpecker, so let me explain what happened there. The red-cockaded woodpecker, *Picoides borealis*, is endangered and nests in forests owned by International Paper (IP). IP and the Fish and Wildlife Service agreed on a target number of breeding woodpecker pairs. Provided that this target number is attained or exceeded, IP will be regarded as complying with the ESA, whatever use they

¹²See for example www.wildlandsinc.com as an illustration of a company in the mitigation banking business.

make of the land (for details, see Bayon 2002, Bonnie 1999, and Bonnie and Bean 1996). Further, the agreement provides that any surplus breeding on the land can be “banked,” that is, used by IP to offset ESA requirements with respect to red-cockaded woodpeckers elsewhere. It is also possible that title to surplus could be sold to other landowners and used by them to gain some measure of exemption. This ability to store or sell the surplus over the amount required by regulations is mitigation banking. As the excess of nesting pairs over a target is saleable, IP now actually has an economic incentive to encourage the endangered species, something it never had with a strict interpretation of the ESA. A few years ago, IP was able to sell banked breeding pairs for about \$100,000 per pair. If several pairs can nest on each acre, this means that the value of land for breeding woodpeckers is greatly in excess of its value as a source of timber.

It is encouraging to see some economic rationality creeping into the ESA,¹³ but it may well be too late for the Act to survive. It would have been so much better if someone had thought about incentive compatibility before the Act was drafted, rather than several decades later.

There are many advantages to market-based policies, and incentive compatibility is certainly one of them. I’m no expert in political economy, but it seems to me that there are political advantages to having a system that works through incentives. It is much harder to work up opposition to a provision that can give people money than to one that prevents them from making it or takes it away from them. If there is such a thing as the political sustainability of legislation, then I think it must come in part from offering options to people rather than taking them away. This is the old lesson about subsidies being hard to remove, but in another form. We should learn from the political durability of subsidies. But the main advantages of a market-based approach are the ones we all know and take for granted—efficiency under the right conditions, cost-effectiveness, and, of course, decentralization. By giving people incentives to do what society wants and then leaving them to choose how to do this, we enlist their knowledge of specific circumstances (as so lucidly explained in von Hayek’s [1945] classic, “The Use of Knowledge in Society”), and also give them an interest in developing better ways of reaching the social goal. The market for SO₂ emissions, established as a part of the 1990 Amendments to the Clean Air Act, illustrates the same point. The costs of removing SO₂ from power station flue gases fell rapidly once the emissions-trading regime was in place, reflecting the incentives established to find less expensive ways of meeting or exceeding target emissions (Carlson et al. 2000). Markets aren’t a panacea, but under the right conditions they work efficiently and with little bureaucracy, and they represent low-maintenance solutions.

Textbooks will tell us that two key shortcomings of markets are their failure to allocate public goods efficiently and the fact that the interests of future generations are not represented. The public good point has to be taken seriously, as so many ESS are public goods. But the cap and trade systems that we have discussed, both for CO₂ under Kyoto and SO₂ under the 1990 Clean Air Act Amendments, are set up to deal with public goods. Atmospheric quality, after all, is unambiguously a public good. It is interesting to understand what has happened here. We have not solved the fundamental problem of incentive compatibility in the provision of public goods: a cap and trade mechanism leaves the decision about the level at which a public good is to be provided to the political process. This is the cap, and it

¹³The NGO Environmental Defense played a major and constructive role here.

is the fundamental resource-allocation decision where incentive-compatibility issues arise. Once the cap is chosen, we use the market to decide how best to meet it. So the role of the market here is limited, but nevertheless important and constructive. Thus, with cap and trade systems, we are allocating goods with a mix of economic and political mechanisms.

Representing the interests of future generations raises hard issues. Analytically, our profession has thought a lot about this: much of the literature I have mentioned on sustainability and intertemporal welfare economics focuses on defining and striking the right balance between the present and the future. But is this something that we can expect a market to implement, or do we need some other form of governance to manage this well? This is not clear. It seems like a very hard problem. One argument is that the present generation naturally thinks about the interests of their successors, their children and grandchildren. To the extent that this happens, the interests of the future are reflected in current choices. There are also groups that see representing the interests of future generations as an explicit part of their mission: some environmental nongovernment organizations see themselves in this light. An interesting example is the UK's National Trust, whose motto is "For Ever, For Everyone." The National Trust's aim is to conserve land and property of outstanding value "For Ever, For Everyone," that is to acquire and hold and manage it so that it is broadly accessible and will be preserved in perpetuity.¹⁴ It has significant purchasing power and can act to buy and to receive gifts, and has been effective in its stated aim of representing the future. At a more empirical level, several authors, such as Kay and Mirrlees (1975), have noted that in the history of industrial societies, the present has never obviously short-changed the future: each generation has been better off than its predecessors. Investment in physical capital and intellectual capital always seems to have passed on to the next generation a capacity to exceed current living standards. Progress has been continuous and cumulative. But there are counterexamples outside the world of industrial societies, as Jared Diamond's (2005) book *Collapse* illustrates. And the central concern in the sustainability literature is that as the stresses we impose on the environment grow almost exponentially, the past may not be a good guide to the future.

Fisheries Management

The last topic I want to talk about is fisheries. This is a very frustrating topic: there is probably no other area of environmental and resource economics where there is so great a gap between real-world performance and what might be achieved if our recommendations were followed. The statistics on the state of the world's fisheries are intensely depressing. Sensible estimates suggest that the biomass of fish in the world's oceans is today perhaps one tenth of what it was half a century ago, and is declining.¹⁵ Even if there is a significant margin of error in this estimate, it is shocking. We are extracting fish from the seas at a rate greatly in excess of the reproduction rate. The stock is falling, and this has been so for a long time. No model of optimal fisheries management would support this.

There are some shortcomings in the standard models of fisheries management, and we need to work on these. Perhaps the main shortcoming is that they model the management

¹⁴See its web site at <http://www.nationaltrust.org.uk/main/>.

¹⁵For a survey see the Report of the Pew Oceans Commission available at www.pewoceans.org. See also Worm et al. (2006).

of a unique species in isolation, whereas in reality fish are part of complex ecosystems, with their dynamics being governed to a large degree by the predator-prey interactions. Typically, these interactions are omitted from our models.¹⁶ This suggests that our models may have been overly generous in predicting acceptable take levels, but the sad fact is that actual levels have been vastly in excess of what our models suggest as reasonable. This is easy to see: with a long-run optimal policy, stocks should be constant, whereas they have fallen very significantly.

It is in fisheries more than any other area that one has to confront the political limitations on adopting rational economic policies. The difficulties here seem to be even greater than with climate change, which, for all the deliberate obstruction and obfuscation from some energy companies, is nevertheless firmly on the world's agenda. Fishing industries are able to block sensible policies every time they are considered, and few governments feel a need to implement even the limited fisheries protection measures that they have in place. Within the last few years, two major commissions, the Pew Oceans Commission and the United States Commission on Oceans Policy, have both written extensive and persuasive reports indicating the need for major reforms to U.S. fisheries management legislation and institutions,¹⁷ yet until very recently none of their recommendations had been adopted.¹⁸ Given the importance of fish as a source of protein, especially in developing countries, the long-term human consequences of current policies could be very costly indeed.

Final Reflections

Many environmentalists, when asked how to resolve our pressing environmental problems, reply that we need a new vision of the relationship between humans and nature. Only with a radical new vision of how we relate to the natural world can we save the planet from the forces of modern technology and economic growth, they argue. Few go on to offer any persuasive details of this new relationship, except to talk about renewable energy, greater energy efficiency, less consumption, and perhaps a new appreciation of the value of nature. Because of the lack of specificity, these visions, while alluring to some, rarely have much practical impact.

My concluding point is that if we want a practical vision of a new and more harmonious relationship between humans and nature, we can find it in environmental economics. Just imagine, for a moment, a world in which all of our recommendations are in place. All external effects are internalized. The importance of natural capital and the services it provides are recognized and feature in national income accounts and in public decision-making. The public good nature of many environmental services is acknowledged, and institutions are in place to manage their provision. Adequate weight is given to the interests of future generations through the roles of interest groups and the selection of discount rates.

In such a world, there really would be a harmonious interaction between human society and the rest of the natural world. Environmental problems, in the sense in which we in economics think of them, would be solved. Our field provides a blueprint for the harmonious

¹⁶ An exception is Carson et al. (2006).

¹⁷ www.pewoceans.org and <http://www.oceancommission.gov/>.

¹⁸ In December 2006 Congress reauthorized the Magnuson Stevens Act and included in the new bill some of the recommendations of the two commissions.

relationship with the natural world that so many wish for yet are not able to describe in any detail. There is a powerful vision behind environmental economics, a vision that can resolve the environmental tensions and problems that concern so many. We need to articulate this more clearly.

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To Tax or Not to Tax: Alternative Approaches to Slowing Global Warming

William D. Nordhaus*

How can countries best coordinate their policies to slow global warming? This study reviews different approaches to the political and economic control of global public goods such as global warming. It compares quantity-oriented mechanisms like the Kyoto Protocol with price-type control mechanisms such as internationally harmonized carbon taxes. The analysis focuses on such issues as the relationship to ultimate targets, performance under conditions of uncertainty, volatility of induced carbon prices, the inefficiencies of taxation and regulation, potential for corruption and accounting finagling, and ease of implementation. It concludes that price-type approaches such as carbon taxes have major advantages for slowing global warming.

Before discussing different approaches, it will be useful to sketch the scientific basis for concerns about global warming. As a result of the buildup of atmospheric greenhouse gases (GHGs), it is expected that significant climate changes will occur in the coming decades and beyond. The major industrial GHGs are carbon dioxide (CO₂), methane, ozone, nitrous oxides, and chlorofluorocarbons (CFCs). Using climate models as well as examining past climate variations, scientists expect significant climatic changes in the coming years. Current estimates are that an increase that doubles the amount of CO₂ or the equivalent in the atmosphere compared with preindustrial levels will, in equilibrium, lead to an increase in the global surface temperature of 1.5–4.5°C, an increase in precipitation and evaporation, and a rise in sea levels of 10–90 cm over this century. Some models also predict regional shifts, such as hotter and drier climates in midcontinental regions, such as the U.S. Midwest. Climate monitoring indicates that the predicted global warming is occurring in line with scientific predictions.¹

While scientists have been analyzing global warming for more than half a century, nations took the first formal steps to slow global warming only about fifteen years ago, under the United Nations Framework Convention on Climate Change (FCCC). The first binding international agreement on climate change, the Kyoto Protocol, came into effect in

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¹Extensive discussions on this subject are contained in reports by the Intergovernmental Panel on Climate Change, especially IPCC 2001, with evidence for recent warming in Hansen et al. (2006).

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2005, and the first period for emissions reductions, 2008–2012, is at hand. The framework for implementing the protocol is most solidly institutionalized in the European Union's Emissions Trading Scheme (EU ETS), which covers almost half of Europe's CO₂ emissions (EU ETS 2006; Klepper and Peterson 2005).

Notwithstanding this apparent success, the Kyoto Protocol is widely seen as a troubled institution. Early problems appeared with the failure to include the major developing countries, the lack of an agreed-upon mechanism to include new countries, and an agreement that is limited to a single period. The major blow came when the United States withdrew from the treaty in 2001. Whereas 65 percent of the 1990 world emissions were included in the original protocol, that number had declined to 32 percent in 2002 with the withdrawal of the United States and strong economic growth in noncovered countries, largely the developing nations of the world. Strict enforcement of the Kyoto Protocol is likely to be observed primarily in those countries and industries covered by the EU ETS. These emissions today account for about 8 percent of global emissions. If the current protocol is extended at the current reduction rates, models indicate that it will have little impact on global climate change (Nordhaus and Boyer 1999; Manne and Richels 1999; Nordhaus 2001; MacCracken et al. 1999).

Nations are now beginning to consider the structure of climate-change policies for the period after 2008–2012. Some countries, states, cities, companies, and even universities are adopting their own climate-change policies. Most global-warming policies adopted by U.S. states or considered by the U.S. federal government contain some mixture of emissions limits and technology standards. Is the Kyoto Protocol a viable long-term approach to this long-term problem? Are there alternatives that might reduce global warming more efficiently? I consider these questions in this article.

The first section describes the political and economic issues raised by global public goods like global warming. It suggests that attempts to coordinate sovereign governments raise thorny problems beyond those involved in most national pollution problems. The next section describes the major mechanisms available to coordinate dealings with global public goods. The subsequent section describes the three fundamental issues that must be resolved in any regime to address climate change—the level of emissions reductions, the distributions of emissions reductions across countries, and the mechanisms to encourage participation of low-income countries. I then describe the price-type approach of harmonized carbon taxes. The penultimate section discusses specific concerns such as how well different approaches meet ultimate objectives, the problem of setting baselines for prices and quantities, treatment of uncertainty in different approaches, the potential for great volatility in the market prices of carbon under quantitative systems, public finance questions, problems of corruption, and administrative issues. I close with a summary of the major issues.

Policies for Global Public Goods

Global warming is a member of a special type of economic activity known as *global public goods*. These are economic or other activities whose impacts are indivisible and whose influences are felt around the world rather than affecting one nation, town, or family. These are not new phenomena. However, they are becoming increasingly prevalent because of rapid technological change and the rapid decline in transportation and communication

costs. What makes global public goods different from other economic activities is that there exist only weak economic and political mechanisms for solving these issues efficiently and effectively.

Dealing with global public goods has been an increasingly important feature of international relations. Aside from global warming, important examples are nuclear proliferation, infectious diseases, intellectual property rights, international trade in goods and services, macroeconomic stability, fisheries, endangered species, and transnational terrorism. We have only to think about recent crises such as those involving weapons of mass destruction, the AIDS epidemic, international financial crises, and the threat of avian flu to realize how prevalent global public goods are. A little further reflection will indicate that nations have had only modest success in agreements to deal with global public goods. There are but a few examples of regimes that manage international public goods effectively, such as those managing international trade disputes (today primarily through the World Trade Organization) and the CFC protocols.

There are major governance issues involved in dealing with global public goods. It is necessary to locate decision making at a level in the hierarchy between the household and a weak or nonexistent world government that can efficiently coordinate solutions. This is a particularly thorny problem for global public goods because global coordination is required. The need for global decision making leads to the Westphalian dilemma. Under international law as it developed in the 1648 Treaty of Westphalia and evolved in Western Europe, obligations may be imposed on a sovereign state only with its consent. Because of the structure of international law, therefore, there is no legal mechanism by which disinterested majorities or even super majorities of countries can coerce noncooperating countries to provide for global public goods. In other words, the Westphalian system is one that allows free-riding. Therefore, we must take entirely different approaches to global public goods compared with those taken for regional, national, or local public goods.

Economic and Focal Public Goods

Looking at the varieties of global public goods, I want to focus on those I will call economic public goods. These activities involve huge numbers of economic agents in a large number of countries, in which the costs and benefits of action do not indicate any obvious focal policy or technological fix. By contrast, I denote as focal public goods those activities in which good policies appear obvious or consensual to most people; for example, it does not take much persuasion to convince people that a reasonable standard is zero AIDS, zero smallpox, zero financial collapses, zero nuclear meltdowns, or zero nuclear explosions.

With economic public goods, by contrast, it is difficult to determine and reach agreement on efficient policies because they involve estimating and balancing costs and benefits where neither is easy to measure and both involve major distributional concerns. Economic public goods include fisheries (where the point of overfishing is difficult to calculate), pollution (where zero pollution is prohibitively expensive), and global warming (where it is apparent that the optimal abatement is today somewhat short of 100 percent of GHG emissions). There is a temptation to redefine economic public goods as focal public goods because that tremendously simplifies analysis and policy. For example, policies have pretended to adopt a complete phase-out of CFCs in principle, although that is impossible in practice. Similarly,

policies to prevent the extinctions of species generally avoid the vexing question of how to draw the line between species and subspecies as well as the intractable question of how far to lower the probability of extinction given that it clearly can never be zero.

Specific Mechanisms to Deal with Global Public Goods

Nations have forged a variety of frameworks for dealing with global public goods and other transnational issues, employing a wide variety of instruments or techniques (Barrett 2003). A partial list is (i) noncooperative, market-based, or laissez-faire approaches (as is currently the case for production of most goods and services as well as for some potential global issues such as asteroid defense); (ii) aspirational or hortatory agreements that urge countries to undertake actions (e.g., the FCCC) or nonbinding voluntary agreements (e.g., the institutional regime created in the 1980s to clean up pollution in the North Sea); (iii) specific and binding treaties, legal contracts among sovereign nations, which are the standard way to deal with international issues (currently in effect for CFCs and many other global environmental agreements); (iv) agreements embedded in broader international institutions or agreements (exemplified when Western nations forced developing countries to accept strong patent protection under the last multilateral trade negotiations); and (v) limited delegations of regulatory or fiscal authority to supranational bodies (seen in some European activities such as the European Central Bank, in some powers of the World Trade Organization [WTO], and in some the international financial institutions). This array of international institutions reminds us that although global warming is a new problem, the problems of international political economy raised by global warming are quite ancient.

When dealing with economic public goods like global warming, it is necessary to reach through governments to the multitude of firms and consumers who make the vast number of decisions that affect the ultimate outcomes. There are two major mechanisms that can be employed—quantitative limits through government fiat and regulation, and price-based approaches through fees, subsidies, or taxes.²

In the global-warming context, quantitative limits set targets on the time path of GHG emissions of different countries. Countries then can administer these limits in their own fashion, and the mechanism may allow transfer of emissions allowances among countries, as is the case under the Kyoto Protocol. This approach has limited international experience under existing protocols such as the CFC mechanisms and broader experience under national trading regimes such as the U.S. SO₂ allowance-trading program.

The second approach is to use harmonized prices, fees, or taxes as a method of coordinating policies among countries. This approach has no international experience in the environmental area, although it has considerable national experience for environmental markets in such areas as the U.S. tax on ozone-depleting chemicals. On the other hand, the use of harmonized price-type measures has extensive international experience in fiscal and trade policies, such as with the harmonization of taxes in the EU and harmonized tariffs in international trade.

²This distinction is drastically simplified. For a nuanced discussion including variants and hybrids, see Aldy, Barrett, and Stavins (2003) and the many references and proposals therein.

Major Issues in Any International Climate-change Regime

Any climate-change regime must face three fundamental issues—the overall level and trajectory of emissions reduction (reflected in a control rate or a market price of carbon emissions), the distribution of emissions reductions across countries, and the need for mechanisms to encourage participation of low-income countries and other reluctant countries. Each of these issues is very contentious.

The Overall Level and Trajectory of Emissions Reduction

Because global warming is a global public good, the key environmental issue is global emissions, and the key economic issue is how to balance costs and benefits of global emissions reductions. Climate change depends only upon total GHG emissions and the time path of emissions, not on the geographic location of emissions. Moreover, the impacts depend primarily upon cumulative emissions that remain in the atmosphere, not on the annual flow of emissions.

Under a price approach, the level of emissions is determined indirectly by the level of the tax or penalty levied on carbon emissions. Under a quantitative approach, the level of emissions is directly chosen. However, a market economy is likely to develop markets for emissions permits, and a market price will therefore emerge. An economist will naturally examine the price in either case, and the first issue can be rephrased as: What is the level of the market price of carbon emissions that is consistent with the regime?

A quantitative measure of the tightness of emissions controls is the value of the “carbon price,” “carbon tax,” or the “social cost of carbon.” The carbon price measures the market price attached to the right to emit 1 ton of carbon through burning fossil fuels or other activities. For calibration purposes, if a hundred-dollar-per-ton carbon tax were to be levied on gasoline, that would raise the price by twenty cents per gallon.³

The key economic question under any regime is whether the price is likely to be high or low. We can examine these questions quantitatively using computerized models built to study the economics of global warming. These models are called “integrated-assessment models,” or IAMs. The use of IAMs has blossomed over the last two decades, and there are now dozens of global models and even larger numbers of models that apply to individual countries.

Carbon prices in efficient emissions reductions

The question of the “optimal” level of emissions reductions is undoubtedly the most difficult and controversial question in the economics of global warming. In a series of studies, my coauthors and I have estimated cost and damage functions and estimated “optimal” or efficient emissions reductions to slow global warming. The results discussed here use the “RICE model” (regional integrated model of climate and the economy), which

³Scientists and economists have customarily measured carbon prices in terms of carbon weight, and I follow that convention. Current emissions-trading programs generally quote in terms of carbon dioxide weight, which has a mass 3.67 times that of carbon. To convert from the carbon units to the CO₂ units, multiply the mass or divide the price by 3.67.

is an IAM that analyzes the major economic trade-offs involved in global warming. It uses the framework of economic growth theory and incorporates emissions and climate modules to analyze alternative paths of future economic growth and global warming.

The latest calculation in the deterministic aggregate RICE model suggests that a 2010 carbon price of around \$17 per ton carbon in 2005 prices—rising to \$70 per ton in 2050—would efficiently balance the costs and benefits of emissions reductions, that is, maximize the present discounted value of benefits minus costs.

It must be recognized that this estimate of the efficient carbon tax is unlikely to capture all the nonmarket aspects of global warming (such as effects on ecosystems), problems of uncertainty and risk aversion, and the potential for “dangerous interferences” with many global processes.⁴ Nonetheless, it does describe a path that recognizes that countries care about their economic development as well as future costs of global warming.

Many other estimates exist for the appropriate market prices of carbon. At the high end is the social cost of carbon proposal in the UK government’s Stern Review (2006) of \$310 per ton of carbon; the very bottom of \$0 is implicit in the policies of global-warming skeptics and the environmental skeptics in the G.W. Bush Administration. However, the relatively low current efficient market price of carbon found in the RICE model was one of the major conclusions in a review of IAMs: “Perhaps the most surprising result is the consensus that given calibrated interest rates and low future economic growth, modest controls are generally optimal” (Kelly and Kolstad 1999).

Emissions reductions and carbon prices in the Kyoto Protocol

Several studies have estimated the economic impacts of the Kyoto Protocol. Modeling estimates indicate that global emissions under the Kyoto Protocol as actually operating would be very close to a “no-controls” baseline, that is, a world without policy-induced GHG emissions reductions. Estimates from the RICE model indicate that global CO₂ emissions in a no-controls world would grow by about 27.5 percent between 1990 and 2010, whereas under the current version of the Kyoto Protocol global emissions growth over the same time period would be around 26 percent. In other words, the analysis indicates that global emissions in 2010 would be 1.5 percent lower than without controls (Figure 1).

Moreover, the RICE model and other studies estimated that the Kyoto Protocol would lead to highly differentiated prices and therefore to an inefficient allocation of abatement across countries (Manne and Richels 1999, 2001; MacCracken et al. 1999; Nordhaus and Boyer 2000; Nordhaus 2001). With the U.S. withdrawal from the protocol, global emissions reductions and carbon prices are projected to be much lower than in the original version. RICE model results indicate that the carbon price in 2010 would be \$41 per ton of carbon with the United States and \$18 without the United States. With the United States out of the picture, the price of permits in Europe would be dramatically lower because the required emissions reductions for the participants would be much smaller.

⁴This term is motivated by the FCCC, which states, “The ultimate objective of this Convention . . . is to achieve . . . stabilization of greenhouse gas concentrations in the atmosphere at a level that would prevent dangerous anthropogenic interference with the climate system.” (United Nations Framework Convention on Climate Change at <http://unfccc.int/2860.php>).

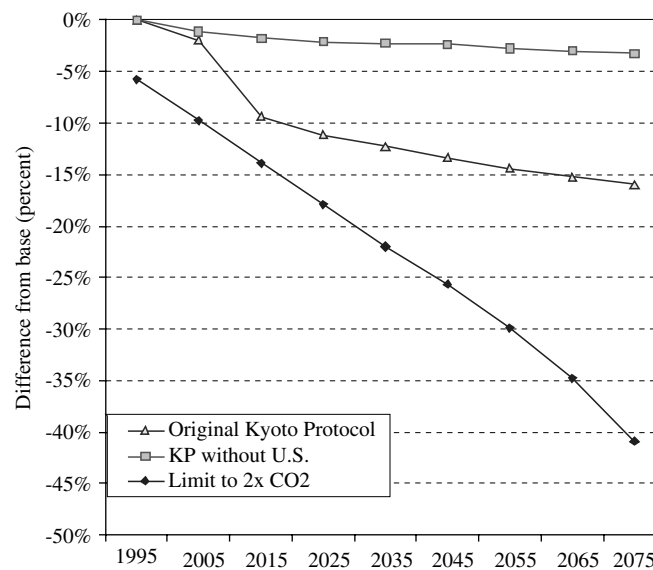


Figure 1. Estimated emissions reductions for different regimes. Numbers are for total global industrial CO₂ emissions and measure the percentage reduction relative to a baseline path of no emissions reductions. The Kyoto paths are “Kyoto forever” and assume that countries freeze their emissions at the 2008–2012 average after the first period with no extension in participation. The “Original Kyoto Protocol” shows the impact of the protocol with United States participation. “KP without United States” shows the impact of removing the United States from the protocol. The “Limit to 2× CO₂” shows the emissions reductions that would minimize the discounted costs of limiting CO₂ concentrations to double preindustrial concentrations. The estimates are for the decades centered on the year shown on the horizontal axis. Source: Nordhaus 2001.

The Distribution of Emissions or Emissions Reductions among Countries

What should be the distribution of emissions reductions among countries, and how should the costs be allocated? These questions apply to differences among high- and low-income countries, among high- and low-emitting countries, and among high- and low-vulnerability countries.

Economics offers a simple, unambiguous, but elusive answer: emissions reductions should be carried out in the most efficient way; and the burden of reducing emissions should be shared in a fair way. The first half of this statement refers to the distribution of actual emissions reductions (discussed in this section), while the second half refers to sharing the costs among countries (which is discussed in the next section).

Emissions reductions will be efficient or “cost effective” if the marginal costs of emissions reductions are equalized across space and, with appropriate discounting, across time. The spatial component of efficiency (“where efficiency”) is that the marginal cost of reductions should be equalized across all countries, industries, and sources. The temporal component (“when efficiency” or intertemporal efficiency) is more complicated. When efficiency requires that the profile of emissions be timed to attain the ultimate goal (whether the goal be concentrations or temperature stabilization or a dynamic cost–benefit criterion). In simple dynamic models, intertemporal efficiency requires that the market price of carbon (equal to the marginal cost of emissions reductions) grows over time at a rate equal to the

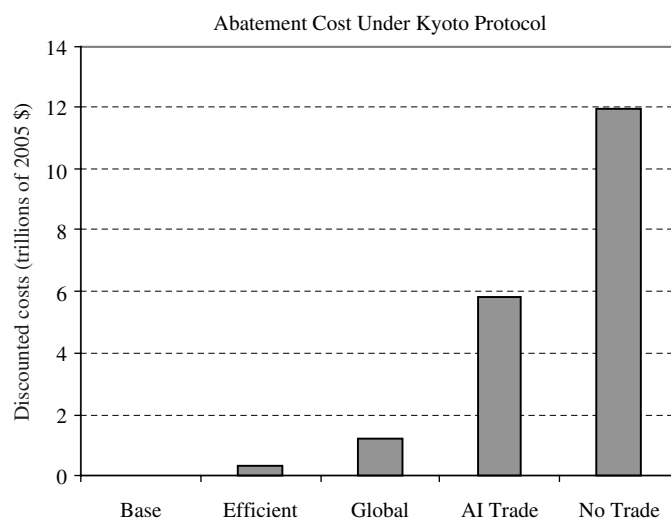


Figure 2. Estimated abatement costs for different implementation strategies of the Kyoto Protocol. This figure shows the discounted value of the costs of abatement and excludes any environmental benefits. Annex I includes high-income countries plus the “transition” economies of Eastern Europe and the former Soviet Union. Costs are discounted to 2005 and are in 2005 U.S. dollars. The *Base* case is with no restraints on emissions and is by definition zero. The *Efficient* case maximizes discounted net benefits, including environmental benefits. *Global* is the case where the emissions under the Annex I countries of the Kyoto Protocol are freely traded among all countries. *All trade* is the basic Kyoto Protocol with full trading of emissions allowances among Annex I countries only. *No Trade* allows no emissions trading among the four major regions of Annex I. The underlying model is described in Nordhaus and Boyer 2000, updated in Nordhaus 2001.

“real carbon interest rate,” which is approximately equal to the real interest rate less the disappearance rate of CO₂ from the atmosphere.

The Kyoto Protocol is defective on both spatial and temporal efficiency criteria because it omits a substantial fraction of emissions (thus failing the spatial criterion) and has no plans beyond the first period (thus failing the temporal dimension of the cost-effectiveness criterion). The two largest emitters (the United States and China) are not included in the current protocol. Figure 2 shows the most recent estimates of abatement costs under different trading regimes for the original Kyoto Protocol using the RICE model. Because it limits trading to a small part of the world and ignores the intertemporal dimension, the Kyoto Protocol is an extremely costly treaty and makes only modest progress in slowing global warming.

Mechanisms to Encourage Participation

How should the economic burden of reducing emissions be shared among countries? “All politics are distributional,” is a maxim of American politics. This is no less true of the politics of international environmental agreements. Neither science nor economics can provide a “correct” answer to the question of how to share the burden of reducing emissions. Disinterested observers might argue that the costs should be allocated on the basis of ability to pay, with richer countries and generations paying a larger fraction of the costs. Interested

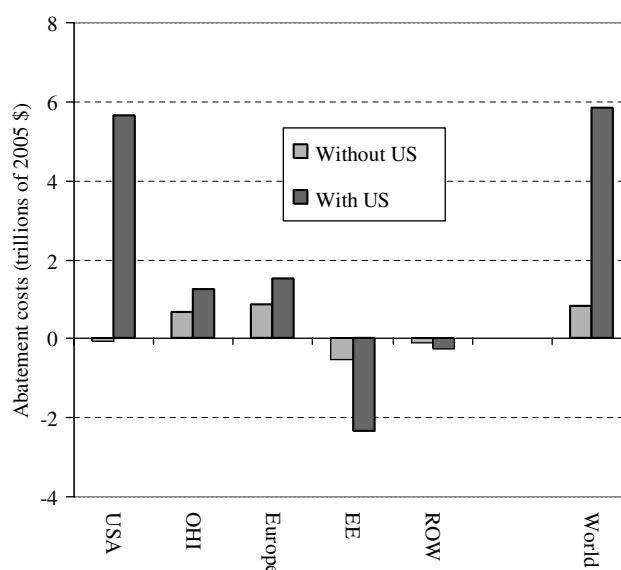


Figure 3. Abatement costs of Kyoto Protocol with and without U.S. participation. This figure shows the discounted value of the costs of abatement (emissions reductions) and excludes any environmental benefits. The first four regions are participants and the last region (ROW) sums the impacts for all non-participants. The burden of abatement shifts greatly with the U.S. withdrawal from the Kyoto Protocol. Source: Nordhaus 2001. OHI, other high-income countries, including Japan and Canada; Europe, primarily the EU; EE, Eastern Europe and the countries of the former Soviet Union; ROW, the rest of the world.

observers, such as negotiating countries, want to pay as little as possible, and are unlikely to participate voluntarily unless they have a positive net benefit.

The Kyoto Protocol has an arbitrary allocation of burdens and transfers because it generally used 1990 emissions as a base year when setting targets in the negotiations during 1997. Consequently, those countries with high emissions in 1990 (such as Russia) are advantaged while those whose emissions have subsequently grown rapidly (such as the United States) are disadvantaged.

The result of the initial allocation is that the Kyoto Protocol was hampered at its inception by a distribution of abatement costs that was weighted heavily toward the United States. Figure 3 shows the RICE model estimates of the costs of abatement for different regions with and without U.S. participation. This study suggested that the United States would bear a large fraction of the costs of implementing the protocol. Indeed, the estimated net benefits for the United States, including environmental benefits (not shown), were negative. At the other extreme, it seems likely that Russia was induced to participate because it would have excess allowances that could be exported.

Just as thorny are questions involving the participation of low-income countries. Efficiency requires full participation of low-income countries in emissions reductions; indeed some of the most economical emissions reductions can be found in low-income countries. But including low-income countries is challenging for many reasons—fairness, development priorities, and pure bargaining strategies. In any mechanism, it seems likely that high-income countries will provide financial and technical assistance to low-income countries to induce participation.

In both the FCCC and the Kyoto Protocol, developing countries were excluded from any obligations to reduce emissions. This approach was probably a fundamental mistake. It is crucial to have a mechanism whereby countries “graduate” into a set of obligations that are commensurate with their abilities to pay—in a way that is similar to the “ability to pay” principle of an income tax system. Part of the challenge is designing a fair graduation procedure; another part is overcoming the Westphalian dilemma of inducing countries to participate when graduation day comes.

Price Approaches to Climate Change

Attempts to address climate change through prices rather than quantities have been discussed in a handful of papers in the economics literature (see Cooper 1998; Pizer 1998; Victor 2001; Aldy, Barrett, and Stavins 2003), but much careful analysis remains to be done. I will highlight a few of the details.

For concreteness, I will discuss a mechanism called harmonized carbon taxes (HCT). Under this approach, there are no binding international or national emissions limits. Rather, countries would agree to penalize carbon emissions at an internationally harmonized “carbon price” or “carbon tax.” Conceptually, the carbon tax is a dynamically efficient Pigovian tax that balances the discounted social marginal costs and marginal benefits of additional emissions. The carbon price might be determined by estimates of the price necessary to limit GHG concentrations or temperature changes below some level thought to be “dangerous interference,” or it might be the price that would induce the efficient level of control. Unlike the quantitative approach under the Kyoto Protocol, there would be no country emissions quotas, no emissions trading, and no base period emissions levels. Because carbon prices would be equalized, the approach would be spatially efficient among those countries that have a harmonized set of taxes. If the carbon tax trajectory follows the rules for “when efficiency,” it would also satisfy intertemporal efficiency.

Details about burden sharing would require study and negotiations. It would be reasonable to allow participation to depend upon the level of economic development. For example, countries might be expected to participate fully when their incomes reach a given threshold (perhaps \$10,000 per capita), and poor countries might receive transfers for early participation. The issues of sanctions, the location of taxation, international trade treatment, and transfers to developing countries under an HCT are important details that require discussion and refinement. If carbon prices are equalized across participating countries, there will be no need for tariffs or border tax adjustments among participants. I emphasize that much work would need to be done to flesh out these arrangements, but they are familiar terrain because countries have dealt with problems of tariffs, subsidies, and differential tax treatment for many years. Some of the thorny administrative issues are discussed below.

Hybrid Approaches

The literature on regulatory mechanisms entertains a much richer set of approaches than the polar quantity and price types that are examined here. An important variant is “prices in quantity clothing”—putting ceilings on the price of emissions-trading permits by combining a tradable permit system with a government promise to sell additional permits

at a specified price (McKibbin and Wilcoxon 2002; Aldy, Barrett, and Stavins 2003). Price caps were considered and rejected by the Clinton Administration in its preparation for the negotiations on the Kyoto Protocol.

The present analysis focuses on the pure strains of the two systems to keep the analysis within manageable limits. From a practical point of view, mixed systems sometimes revert to their archetypes. For example, even though the Kyoto Protocol was designed to allow complete trading among the participants, there have been strong pressures to limit trading and force countries to make much of their reductions domestically. The EU implementation of the Kyoto Protocol allows full trading within the EU but limits the purchases of emissions permits from other countries. The lesson from foreign-trade barriers, where price and quantity limits have a much longer history, is that the quantity limits imposed through quotas are extremely durable.

Comparison of Price and Quantity Approaches

The Kyoto Protocol lacks any connection to ultimate economic or environmental policy objectives. Freezing emissions at a given historical level for a group of countries is not related to any identifiable goals for concentrations, temperature, costs, damages, or “dangerous interferences.” Nor does it bear any relationship to an economically oriented strategy that would minimize the costs of attaining environmental and economic objectives.

Price-type systems such as taxes have a mixed record of efficiency. In this context, the ideal system for a harmonized carbon tax is relatively simple, as described above. Because of its conceptual simplicity, it might prove simpler to design an efficient tax than an efficient quantity mechanism.

Setting Baselines for Prices and Quantities

Quantity limits are particularly troublesome where targets must adopt to growing economies, differential economic growth, uncertain technological change, and evolving science. These problems are especially prominent under the Kyoto Protocol, which set its targets thirteen years before the date on which the controls become effective (2008–2012), and used baseline emissions from twenty years before the control period. Base year emissions have become increasingly as obsolete as the economic and energy structures, and even political boundaries of countries have changed.

The baselines for future budget periods and for new participants will present deep problems for extensions of a quantity regime like the Kyoto Protocol. A natural baseline for the post-2012 period would be a no-controls level of emissions. That level is in practice impossible to calculate or predict with accuracy for countries with abatement policies in place. Problems would arise in the future as to how to adjust baselines for changing conditions and to take into account the extent of past emissions reductions.

Under a price approach, the natural baseline is a zero carbon tax or penalty. Countries’ efforts are then judged relative to that baseline. It is not necessary to construct a historical base year of emissions. Countries are not advantaged or disadvantaged by their past policies or the choice of arbitrary dates. The question of existing energy taxes may raise similar complications, and I address these below. Moreover, there is no asymmetry between early

joiners and late joiners, and early participants are not disadvantaged by having their baseline adjusted downward.

Treatment of Uncertainty

Uncertainty pervades climate-change science, economics, and policy. One key difference between price and quantity instruments is how well each adapts to deep uncertainty. A major result from environmental economics is that the relative efficiency of price and quantity regulation depends upon the nature—and more precisely the degree of non-linearity—of costs and benefits (Weitzman 1974). If costs are highly nonlinear compared to benefits, then price-type regulation is more efficient; conversely, if the benefits are highly nonlinear while the costs are close to linear, then quantity-type regulation is more efficient.

While this issue has received scant attention in the design of climate-change policies, the structure of the costs and damages in global warming gives a strong presumption to price-type approaches. The reason is that the benefits are related to the stock of GHGs, while the costs are related to the flow of emissions. This implies that the marginal costs of emissions reductions are highly sensitive to the level of reductions, while the marginal benefits of emissions reductions are close to independent of the current level of emissions reductions (Pizer 1999; Hoel and Karp 2001). This combination means that emissions fees or taxes are likely to be much more efficient than quantitative standards or tradable quotas when there is considerable uncertainty. This insight applies far beyond global warming to any stock public good.

Volatility of the Market Prices of Tradable Allowances

Uncertainties affect prices. Because supply, demand, and regulatory conditions evolve unpredictably over time, quantity-type regulations are likely to cause volatile trading prices of carbon emissions. Price volatility for allowances is likely to be particularly high because of the complete inelasticity of the supply of permits along with highly inelastic demand for permits in the short run.

The history of European trading prices for CO₂ illustrates the extreme volatility of quantity systems. Over 2006, the range of trading prices has been from \$44.47 to \$143.06 per ton carbon (Point Carbon 2006). The prices of allowances fell by more than 70 percent in one month because of new regulatory information.

More extensive evidence with the trading of quantitative environmental allowances comes from the history of the U.S. sulfur dioxide (SO₂) emissions-trading program. This program includes an annual auction conducted by the EPA as well as private markets, in which firms and individuals can buy and sell allowances. The comparison between SO₂ prices and carbon trading prices is useful because of the similar economic characteristics of the respective markets. Both markets are ones in which the supply is fixed or near-fixed in the short run. Moreover, for each market, the demand is highly inelastic because it involves the substitution between a fuel (such as coal) and other inputs, where the technology is relatively inflexible in the short run and substitution is therefore limited. To some extent, the volatility can be moderated if an agreement allows banking and borrowing, meaning that countries can draw from future emissions allowances, or save allowances for the future.

But programs are unlikely to allow borrowing, and banking provides only limited relief from price volatility.

We can gain some insight into the likely functioning of CO₂ allowances by examining the historical volatility of the price of SO₂ allowances. Spot SO₂ prices at the annual EPA auction have varied from a low of \$66 per ton in 1996 to a high of \$860 per ton in 2005. Futures prices have varied by a factor of 4.7 (EPA 2006). If we look at the private market, we find that allowance prices have varied by a factor of 69 in the 1995–2006 period and by a factor of 12 in the 2001–2006 period. Some changes have been induced by changes in regulatory policies, but that feature would be relevant for the carbon market as well.

We can obtain a more precise measure of variability by calculating the statistical “volatility” of the prices of SO₂ emissions allowances and comparing them with other volatile prices. Volatility measures the average absolute month-to-month change, and is a common approach to indicating the variability and unpredictability of asset prices. Figure 4 shows the estimated volatility of four prices for the period 1995–2005: the consumer price index, stock prices, SO₂ allowance prices, and oil prices. SO₂ prices are much more volatile than stock prices (or than the prices of other assets such as houses, not shown); they are vastly more volatile than most consumer prices; and their volatility is close to that of oil prices.

Such rapid fluctuations are costly and undesirable, particularly for an input (carbon) whose aggregate costs might be as great as petroleum in the coming decades. An analogous situation occurred in the United States during the monetarist experiment of 1979–82, when the Federal Reserve targeted quantities (monetary aggregates) rather than prices (interest rates). During that period, interest rates were extremely volatile. In part owing to the increased volatility, the Fed changed back to a price-type approach after a short period of experimentation (Poole 1970). This experience suggests that a regime of strict quantity

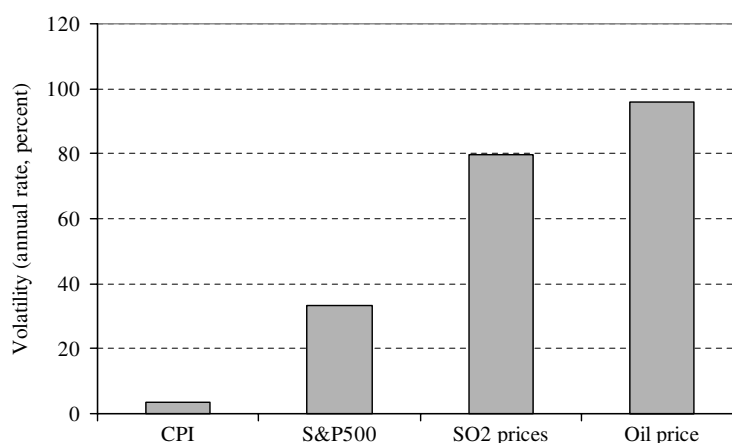


Figure 4. Prices of sulfur emissions allowances show high volatility. This figure shows the estimated volatility of four prices over the 1995–2006 period. These are, from left to right, the consumer price index (CPI), the stock price index for the Standard and Poor 500 (S&P500), the price of U.S. SO₂ allowances (SO₂ prices), and the price of crude oil (Oil price). Volatility is calculated as the annualized absolute logarithmic month-to-month change. Source: Oil prices, CPI, and stock prices from DRI database available at Yale University. Prices of SO₂ permits are spot prices provided by Denny Ellerman and reflect the trading prices.

limits might have major disruptive effects on energy markets and on investment planning, as well as on the distribution of income across countries, inflation rates, energy prices, and import and export values. It might consequently become extremely unpopular with market participants and economic policymakers.

Public Finance Questions

Another important merit of carbon taxes is the strong fiscal-policy advantage of using revenue-raising measures. When tax or regulatory restrictions raise goods prices, this increases inefficiency losses from the existing tax system. The reasoning is that the existing tax and regulatory system raises prices above efficient levels. Adding further taxes or regulations to existing ones increases the inefficiency or “deadweight loss” of the existing system and should be counted as part of the additional costs of global-warming policy. This effect is the “double burden” of taxation, analyzed in the theory of the “double dividend” from green taxes (Goulder, Parry, and Burtraw 1997; Goulder and Bovenberg 1996.)

If the carbon constraints are imposed through taxes, and the revenues are returned by reducing taxes on other goods or inputs, then the increased efficiency loss from taxation can be mitigated, so that there is no net increase in deadweight loss. If the constraints under a quantity-based system are imposed by allocations that do not raise revenues, however, then there is no mechanism to mitigate the increased deadweight loss. This is an important issue, as the inefficiency losses can be as large as abatement costs.

While it is possible that emissions permits will be auctioned (thereby generating revenues with which the tax burden can be mitigated), practice suggests that most of the permits would be allocated at zero cost to “deserving” parties, or distributed to reduce political resistance. In the cases of SO₂ allowances and CFC production allowances, virtually *all* the permits were allocated at no cost to producers. The major conclusion is that using tax approaches rather than quantity-type approaches will help promote a more efficient collection and recycling of the revenues from the carbon constraints.

Rents, Corruption, and the Resource Curse

An additional question, applying particularly to international environmental agreements, concerns the administration of programs in a world in which governments vary in terms of honesty, transparency, and effective administration. Quantity-type systems are much more susceptible to corruption than price-type regimes. An emissions-trading system creates valuable assets in the form of tradable emissions permits and allocates these to countries. Limiting emissions creates a scarcity where none previously existed. It is a rent-creating program. The dangers of quantity as compared to price approaches have been demonstrated frequently when quotas are compared with tariffs in international trade interventions.

Rents lead to rent-seeking behavior. Additionally, resource rents may increase unproductive activity, civil and international wars, and slow economic growth—this being the theory of the “resource curse” (Sachs and Warner 1995; Torvik 2002). The scarce permits can be used by the country’s leaders for nonenvironmental purposes rather than to reduce emissions. Dictators and corrupt administrators could sell part of their permits, and pocket the proceeds.

Calculations suggest that tens of billions of dollars of permits may be available for foreign sale from Russia under the Kyoto Protocol. Given the history of privatizing valuable public assets at artificially low prices, it would not be surprising if the carbon market became tangled in corrupt practices, undermining the legitimacy of the process. We might also imagine a Kyoto Protocol extended to developing countries. Consider the case of Nigeria, which had carbon emissions of around 100 million tons in recent years. If Nigeria were allocated tradable allowances equal to recent emissions and could sell them for \$20 per ton of carbon, this would raise around \$2 billion of hard currency annually—in a country whose nonoil exports in 2000 were only \$600 million.

Problems of financial finagling are not limited to poor, weak, or autocratic states. Concerns arise in the wake of the recent accounting scandals in the United States. A cap-and-trade system relies upon accurate measurement of emissions or fossil fuel use by sources in participating countries. If firm A (or country A) sells emissions (or carbon-content) permits to firm B (or country B), where both A and B are operating under caps, then it is essential to monitor the emissions (or fuel use) of A and B to make sure that their emissions (fuel use) are within their specified limits. Indeed, if monitoring is ineffective in country A but effective in country B, a trading program could actually end up raising the level of global emissions because A's emissions would be unchanged while B's would rise. Incentives to evade emissions limitations in an international system are even stronger than the incentives for tax evasion. Tax cheating is a zero-sum game for the company and the government, while emissions evasion is a positive sum game for the two parties.

A price approach gives less room for corruption because it does not create artificial scarcities, monopolies, or rents. There are no permits transferred to countries or leaders of countries, so they cannot be sold abroad for wine or guns. There is no new rent-seeking opportunity. Any revenues would need to be raised by taxation on domestic consumption of fuels, and a carbon tax would add absolutely nothing to the rent-producing instruments that countries have today.

Administrative and Measurement Issues

One objection to the carbon-tax approach concerns its administration. The issue has been analyzed by David Victor in his analysis of the Kyoto Protocol:

Monitoring and enforcement [of a carbon tax approach] are extremely difficult. . . . In practice, it would be extremely difficult to estimate the practical effect of the tax, which is what matters. For example, countries could offset a tax on emissions with less visible compensatory policies that offer loopholes for energy-intensive and export-oriented firms that would be most adversely affected by the new carbon tax. The resulting goulash of prior distortions, new taxes, and political patches could harm the economy and also undermine the goal of making countries internalize the full cost of their greenhouse gas emissions. (Victor 2001, 86)

Such concerns are serious. The major obstacle to enforcement is the measurement of "net carbon taxes." As Victor notes, we would need to measure net carbon taxes in the context of other fiscal policies (such as fuel taxes and coal subsidies). For example, suppose that Poland imposed a fifty-dollar carbon tax, which would fall primarily on coal. It might at the

same time increase coal subsidies to offset the carbon tax, thereby reducing the level of net carbon taxes. Alternatively, Canada might argue that it has met its carbon-tax obligations by raising provincial stumpage charges on timber. How would the carbon tax be calculated in such circumstances?

One approach would be to calculate the net taxation of carbon fuels, including all taxes and subsidies on energy products, but not to go beyond this to indirect, embodied impacts (i.e., carbon used to produce inputs into production) outside of exceptional cases. Such a calculation would require two steps. First, each country would provide a full set of taxes and subsidies relating to the energy sector; second, we would need an appropriate methodology for combining the different numbers into an overall carbon tax rate. A final issue is how to count initial taxes.

Obtaining data on country tax rates

The first issue—obtaining tax rates—is relatively straightforward for market economies. One of the proponents of the tax approach, Richard Cooper, describes the monitoring issue as follows:

Monitoring the imposition of a common carbon tax would be easy. The tax's enforcement would be more difficult to monitor, but all important countries except Cuba and North Korea hold annual consultations with the International Monetary Fund on their macroeconomic policies, including the overall level and composition of their tax revenues. The IMF could provide reports to the monitoring agent of the treaty governing greenhouse gas emissions. Such reports could be supplemented by international inspection both of the major taxpayers, such as electric utilities, and the tax agencies of participating countries. (Cooper 1998)

Additionally, the levels of taxes and subsidies are generally public knowledge, particularly in market democracies, where they are part of the legislative process. On the other hand, countries with closed political systems might attempt to hide their subsidies. This problem would be particularly troublesome in nonmarket economies or in sectors in which fuels are allocated directly rather than by the price mechanism. Direct allocation is becoming the exception rather than the rule in the world today, however.

Conceptual issues in measuring tax rates

The second issue, calculating the effective carbon tax from the underlying data, is a technical economic issue. Calculations would require conventions about how to convert energy taxes into their carbon equivalent. Some of the calculations involve conversion ratios (from coal or oil to carbon equivalent) that underpin any control system. Others would require input–output coefficients, which might not be universally available on a timely basis. On the whole, calculations of effective carbon tax rates are straightforward as long as they do not involve indirect or embodied emissions.

To go beyond first-round calculations to indirect effects would require assumptions about supply and demand elasticities and cross-elasticities, might engender disputes among countries, and should be avoided if possible. The procedures would probably require mechanisms similar to those used in WTO deliberations, where technical experts would

need to calculate effective taxes under a set of guidelines that would evolve under quasi-legal procedures. Many of these issues are discussed in the literature on ecological taxes (von Weizsäcker and Jesinghaus, 1992).

How to count initial carbon taxes

A final issue involves the question of how to count initial carbon taxes. Some countries—particularly those in Europe—might claim that they already have high carbon-equivalent taxes because of high taxes on gasoline. They would argue for taking existing taxes into account before requiring them to undergo further obligations.

While this looks like a subterfuge, counting pre-existing taxes as compliance is appropriate and is easily seen as such in the carbon-tax framework. From the point of view of global efficiency, it makes no sense for countries with high existing taxes to add further penalties on top of existing ones before countries with subsidies or no penalties impose their carbon taxes. Therefore, the first step, and one absent from analysis of the Kyoto Protocol, would be a calculation of existing equivalent carbon taxes and subsidies. Nordhaus and Boyer calculated that, even without CO₂ taxes, Europe is taxing carbon at a rate of approximately one hundred dollars per ton of carbon more than the United States (Nordhaus and Boyer, 2000). Given that disparity, it would make no economic sense to require Europe to add even higher carbon taxes on top of its existing ones before other countries impose even modest carbon taxes.

Conclusion

We are just beginning to understand and cope with the “great geophysical experiment” of global warming (Ravelle and Seuss, 1957). In this article, I suggest that price-type approaches such as HCTs are more efficient instruments than quantity approaches like those found in the Kyoto Protocol. Under the tax approach, countries set market penalties on GHG emissions at levels that are equalized across different regions and industries. The tax would start relatively low and then, unless the outlook changes for better or worse, rise steadily over time to reflect the increasing prospective damages from global warming.

Many considerations enter the balance in weighing prices and quantities. One advantage of price-type approaches is that they can more easily and flexibly integrate economic costs and benefits of emissions reductions, whereas the approach in the Kyoto Protocol has no discernible connection with ultimate environmental or economic goals. This advantage is emphatically reinforced by the large uncertainties and the evolving scientific knowledge in this area. Emissions taxes are more efficient in the face of massive uncertainties because of the relative linearity of the benefits compared with the costs. A related point is that quantitative limits will produce high volatility in the market price of carbon under an emissions-targeting approach. In addition, a tax approach can capture the revenues more easily than quantitative approaches, and may add less to the distortion caused by existing taxes. The tax approach also provides less opportunity for corruption and financial finagling than quantitative limits, because it creates no artificial scarcities to encourage rent-seeking behavior.

However, we must be realistic about the shortcomings of the price-based approach. It is unfamiliar ground in international environmental agreements. Tax is almost a four-letter word. Many people distrust price approaches in general; they are of special concern for global warming because they do not impose explicit limitations on the growth in emissions or the concentrations of GHGs. We might fear that the international community could fiddle with tax rates and definitions and measurement issues and coverage while the planet burns. These are real concerns and will require time and patience to address and overcome.

The coming years will undoubtedly witness intensive negotiations on global warming as the planet warms, the oceans rise, and new ecological and economic impacts are discovered, especially if threats of abrupt or catastrophic impacts become more likely. A dilemma will arise particularly if, as has been suggested above, the quantitative approach under the Kyoto Protocol proves ineffective and inefficient. As policy makers search for more effective and efficient ways to slow dangerous climatic change, they should consider the possibility that price-type approaches like harmonized taxes on carbon are powerful tools for coordinating policies and slowing global warming.

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Uncertainty in Environmental Economics

Robert S. Pindyck*

Introduction

An introductory course in environmental economics typically teaches students that the design and evaluation of a policy to deal with an environmental problem boils down to cost-benefit analysis. The instructor might proceed as follows. Left to their own, humans (i.e., producers and consumers) do bad things to the environment, such as polluting rivers and lakes, spewing sulfur dioxide into the air, and releasing ozone-depleting chlorofluorocarbons (CFCs). Government intervention—restrictions or taxes on emissions, banning CFCs—prevents some of this destructive behavior, and thereby reduces the amount of environmental damage. But it does so at a cost (e.g., electric power producers must install expensive scrubbers, and air conditioners must be made with a more expensive or less efficient refrigerant). So the policy problem boils down to deciding whether the benefit in terms of less environmental damage is at least as large as the cost of the policy. Of course, the benefits (and often some of the costs) usually occur in the future, and therefore must be expressed in present value terms. So, given a discount rate, it is all quite simple: calculate the present value of the benefits of a policy, subtract the present value of the costs, and see whether the difference (the net present value, or NPV) is positive. And if one is comparing several alternative policies, choose the one with the highest NPV.¹

The student, however, may start to realize that the problem is in fact more complicated:

- First, we never really know what the benefits from reduced environmental damage will be, or even the amount of environmental damage that will be reduced by a particular policy. Worse yet, we *cannot know* with much precision what those benefits will be even if we work very hard to find out. Take the case of global warming. Modern meteorological science tells us that the relationships between greenhouse gas (GHG) concentrations, temperatures (regional or global), and climate patterns are inherently stochastic (i.e., partly random). And even if we knew what those changes

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¹For a good textbook discussion of cost-benefit analysis applied to environmental policy, see Tietenberg (2006).

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in temperatures and climate patterns are likely to be, we know even less about their economic and social impact, in part because we do not know how humans will adapt (e.g., by growing different crops or living in different areas). And if you think global warming is an unfair example because of the very long time horizons involved, take the example of acid rain. Although virtually everyone would agree that the acidification of lakes and rivers—as well as the direct effects on human health—from unregulated nitrogen oxide (NO_x) and sulfur oxide (SO_x) emissions is not a good thing, there is very little agreement as to just how bad it is.

- Second, we usually do not know what the current and future costs of a policy will be. In the case of a carbon tax, for example, we do not know how consumers and producers will respond, especially over the long term. For example, to what extent will consumers use less fuel and buy more fuel-efficient cars and heating systems? And will producers develop and adopt more fuel-efficient technologies? Or in the case of NO_x and SO_x emissions, we know the current cost of scrubbers, but we do not know what their cost will be in the future and how installing them will affect electricity prices and demand.
- Third, what discount rate (or rates) should be used to calculate the present values? There is disagreement among economists regarding the “correct” rate that accounts properly for social time preferences and risk. And even if we settled on a conceptual notion of a “correct” rate, for example, society’s marginal rate of return on capital, there would still be considerable uncertainty over the actual numbers for current and future discount rates. (The marginal return on capital is difficult to measure, and its future evolution is inherently uncertain.) Furthermore, as we will see, discount rate uncertainty is itself a determinant of the “correct” effective rate that should be used for policy evaluation.

Of course, the student might argue that there is nothing problematic about uncertainty over current and future benefits, costs, and discount rates. If the student had taken a basic course on finance, she would know that firms frequently make capital investment decisions in the face of similar uncertainties over the future cash flows from the investment, and must select discount rates subject to uncertainty over the correct risk premium. She might argue that firms typically base their investment decisions on the expected values of those cash flows, and environmental policy designs can likewise be based on expected values.

The student might also argue that most public policy decisions must be made in the face of uncertainty. Possible changes in our Social Security or Medicare programs must be evaluated in the context of a broad set of uncertainties over future changes in the demographic makeup of the country, changes in incomes, savings rates and costs of living for different demographic groups, and changes in disease prevalence and medical costs, to name a few. Likewise, it is notoriously difficult to predict the effects of changes in tax policy on income, employment, and government tax revenues. Again, what is special about environmental policy?

I would counter that for many environmental problems the uncertainties are greater and more crucial to policy design and evaluation. In particular, three important complications arise that are often crucial for environmental policy, but are usually much less important for most other private and public policy decisions.

1. The first complication is that environmental cost and benefit functions tend to be *highly nonlinear*. In other words, the damage likely to be caused by air or water pollution or by GHG emissions does not increase linearly with the level of pollution or emissions. Instead, the damage might be barely noticeable for low levels of pollution and then become severe or even catastrophic once some (uncertain) threshold is reached. Likewise, the cost of pollution abatement may be very low for low levels of abatement but then become extremely high for higher or total abatement. This means that one cannot simply use expected values; the expected value of the cost or benefit function will be very different from the function of the expected value.

Furthermore, the precise shapes of the functions are unknown. This is particularly important if we believe that there is a threshold or “tipping point” at which the impact of a pollutant becomes extremely severe, but we do not know where that point is. For example, how large an increase in GHG concentrations—or in mean temperature—would it take for the consequences to be near-catastrophic? And at what point would over-fishing or habitat destruction lead to the collapse or extinction of a fish or animal population? The lack of answers to these questions suggests that environmental policy should be “precautionary” in the sense of favoring earlier and more intense intervention. But just how “precautionary” should the policy be? Should countries agree, for example, to roll back their GHG emissions to the 1930 levels? I do not mean to be facetious. Rather, I want to stress that uncertainty over the existence and/or position of a “tipping point” can be critical to policy timing and design.

2. The second complication is that environmental policies usually involve important *irreversibilities*, and those irreversibilities sometimes interact in a complicated way with uncertainty. There are two kinds of irreversibilities that are relevant for environmental policies, and they work in opposite directions.

First, policies aimed at reducing environmental degradation almost always impose sunk costs on society. These sunk costs can take the form of discrete investments (e.g., coal-burning utilities might be forced to install scrubbers), or they can take the form of expenditure flows (e.g., a price premium paid by a utility that has committed to burning low-sulfur coal). In either case, if future costs and benefits of the policy are uncertain, these sunk costs create an opportunity cost of adopting the policy, rather than waiting for more information about environmental impacts and their economic consequences. This implies that traditional cost-benefit analysis will be biased toward policy adoption.

Second, environmental damage is often partly or totally irreversible. For example, atmospheric accumulations of GHGs are long lasting; even if we were to drastically reduce GHG emissions, atmospheric concentration levels would take many years to fall. Likewise, the damage to ecosystems from higher global temperatures, acidified lakes and streams, or the clear-cutting of forests may be permanent. This means that adopting a policy now rather than waiting has a sunk benefit, that is a negative opportunity cost. This implies that traditional cost-benefit analysis will be biased against policy adoption.

How important are these irreversibilities and what are their implications for policy? The answers depend on the nature and extent of the uncertainties over costs and

benefits, and how those uncertainties are likely to get resolved over time. The greater the current uncertainties, and the greater the rate at which they will be resolved, the greater will be the opportunity costs and benefits associated with policy adoption.

3. Third, unlike most capital investment projects and most other public policy problems, environmental policies often involve *very long time horizons*. While NPV calculations for firms' investments rarely go beyond twenty or twenty-five years, the costs and especially the benefits from an environmental policy can extend for a hundred years or more. The problems of global climate change and nuclear waste disposal are well-known examples with long time horizons, but there are also others. For some forests and the ecosystems they contain, clear-cutting and other interventions can have consequences that extend for many decades; likewise for chemical contaminations of land or water supplies. And the extinction of a species is, by definition, forever.

A long time horizon exacerbates the uncertainty over policy costs and benefits. It is hard enough to predict the impact of pollution or the costs of abatement five or ten years from now. Over a fifty-year horizon, the uncertainties are much greater.

A long time horizon also makes discount rate uncertainty much more important. Suppose we are not sure whether the "correct" discount rate for evaluating a policy is two percent or four percent. (Many rates have been used in policy analysis; the OMB has suggested 3 percent and 7 percent for government regulatory analyses.) With a 2 percent discount rate, a \$100 benefit fifty years from now is worth about \$37 today, but with a 4 percent rate it is worth only \$14 today. If the \$100 benefit accrues a hundred years from now, its present value is about \$14 with the 2 percent rate, but only \$2 with the 4 percent rate. Clearly, with discount rates of 4 percent or more, it would be very hard to justify almost any policy that imposes costs today but yields benefits only fifty or a hundred years in the future.

Uncertainty over future discount rates has an important implication for the choice of discount rate that we should use in practice—it makes that rate *lower* than any expected future discount rate. The expected present value of \$100 received T years from now is the expected value of $\$100 / (1 + R)^T$, where R is the (uncertain) discount rate (and $1 / (1 + R)^T$ is called the *discount factor*). But the expected value of $\$100 / (1 + R)^T$ is *greater than* $\$100 / (1 + R_e)^T$, where R_e is the expected value of R . Furthermore, the longer the time horizon (i.e., the larger the T), the greater is the difference between the expected value of $\$100 / (1 + R)^T$ and $\$100 / (1 + R_e)^T$. This means that the effective discount rate that should be used in a present value calculation is *less than* the expected (or average) discount rate.

The remainder of this article discusses the sources and nature of the uncertainties that tend to rise in environmental economics, and how the policy implications of those uncertainties are shaped by the nonlinear nature of benefit and cost functions, by the irreversibilities that are often present, and by long time horizons. The next section explains why benefits and costs are inherently uncertain, and discusses the nonlinear characteristics of environmental benefit and cost functions. I focus in particular on uncertainty over possible "tipping points," that is a threshold resulting in catastrophic environmental damage. The following two sections deal with irreversibilities and the opportunity costs and benefits they create and examine the implications of long time horizons and discount rate uncertainty. The article

concludes by summarizing the lessons for policy design, and the areas where much more research is needed.

Despite its title, this article is not intended to be a comprehensive survey of the many aspects of uncertainty in environmental economics, or the vast amount of recent and ongoing research on the topic. This is, after all, an article, not a book, and I have chosen to focus on those areas where I have a research interest, and where I believe some elucidation would be worthwhile.

The Uncertain Nature of Benefits and Costs

I claimed in the previous section that, in comparison to many other public policy problems, environmental problems typically involve uncertainties that are greater and more crucial to policy design and evaluation. Why? As I have already explained, and will discuss in more detail later, environmental policy design must contend with highly nonlinear benefit and cost functions, irreversibilities, and long time horizons. But in addition, environmental problems usually involve three compounding levels of uncertainty—uncertainty over the underlying physical or ecological processes, uncertainty over the economic impacts of environmental change, and uncertainty over technological changes that might ameliorate those economic impacts and/or reduce the cost of limiting the environmental damage in the first place. (By “economic impacts,” I mean to include health impacts, lost consumer and producer surplus from degraded air, water, fisheries, and other public goods, as well as lost output resulting directly from changes in climate, resource availability, etc.)

These compounding levels of uncertainty apply to both the benefit and cost sides of policy design and evaluation. For example, the future benefits of reducing GHG emissions depend first on the uncertain relationships between GHG emission levels, GHG concentrations, and resulting temperature distributions (as well as other climatic effects). But those benefits also depend on the uncertain economic impacts of changes in temperature distributions, as well as adaptation to climate changes (e.g., the use of new plant varieties or irrigation methods) that might reduce those economic consequences. The current and future costs of reducing anthropogenic climate change depend on the amount by which GHG emissions would need to be reduced, but that in turn depends on the uncertain relationships between GHG emission levels, concentrations, and climate effects, and the uncertain economic impacts of the climate effects. (It is the economic impacts, broadly construed, that is the policy concern in the first place.) And those costs also depend on unpredictable technological advances in energy conservation, cheaper nonpolluting energy sources, and methods of carbon sequestration.

These uncertainties and their effects on policy are magnified by the often nonlinear nature of benefit and cost functions. Uncertainties over benefits and costs are manifested not only in the form of parameter uncertainty (e.g., uncertainty over the elasticity of emissions with respect to a tax rate on emissions), but also in the form of uncertainty over the shapes of the (nonlinear) benefit and cost functions (e.g., uncertainty over how that elasticity falls as the tax rate is increased). As discussed below, the problem becomes especially severe when there are “tipping points” (i.e., when at some level of environmental damage the consequences become near-catastrophic), but we do not know what that point is.

To make the discussion over benefits and costs more concrete, I will focus on global warming, for which the policy implications of uncertainty have been studied extensively (but

certainly not exhaustively). Later I will turn to the nature and implications of uncertainty for a very different environmental policy problem—the regulation of a fishery.

Uncertainty over Benefits

The point of environmental policy is to bring human exploitation of environmental assets closer to socially optimal levels, thereby creating social benefits. We might impose a carbon tax to reduce the future economic impact of global warming because we expect the benefits (a reduced economic impact from reduced warming) to outweigh the costs of the policy. But how soon should we impose a carbon tax, and how large should it be? Putting aside costs for the moment, the answers depend on the specific benefit functions, that is on how the benefits from the tax vary with its size. And the answers also depend on the nature and extent of uncertainty over those benefits.

The benefits over the next hundred years from reducing GHG emissions depend on (1) expected GHG emission levels absent abatement; (2) how rapidly atmospheric GHG concentrations will grow at the given emission levels; (3) how higher GHG concentrations will affect global temperatures; and (4) how large an economic impact we should expect from higher temperatures.² The uncertainties are substantial for each step in this chain. Combining the first three steps, if we take the Intergovernmental Panel on Climate Change (IPCC) report (2001) as a benchmark, the projected increase in mean temperature by the end of this century absent an abatement policy ranges from 1.4°C to 5.8°C. This large range should not be surprising. GHG emission levels are hard to predict because they depend on uncertain economic growth and energy intensities, and energy intensities in turn depend on unpredictable changes in energy prices and technologies.³ And even if emission levels were known, changes in GHG concentrations and temperature depend on a complex physical system (with feedback loops that might be positive or negative) that is poorly understood.

The economic impact of global warming is even harder to predict. A temperature increase in the middle of the IPCC range—say, 2.5°C—would likely be beneficial for some regions of the world (e.g., Russia and Canada), harmful for other regions (e.g., India, Bangladesh, and Southeast Asia), and might have little or no aggregate net impact on other regions (e.g., the United States). Given the long time horizon involved, we would expect people to adapt to the changing temperatures and changing climates generally (e.g., by planting and developing different crops, and by moving from warm and low-lying areas to cooler and higher ones). Studies have shown that potential adaptation is clearly important (e.g., Mendelsohn, Nordhaus, and Shaw [1994]), but it is still difficult to predict to what extent people will adapt and at what cost.

²I am simplifying the problem for expositional purposes. For example, GHGs include methane as well as CO₂, and the mix matters. Likewise, it is not simply the change in mean temperature that matters, but also the geographic distribution of temperature changes. And, of course, economic impacts can be complex and vary substantially across geographical regions.

³Kelly and Kolstad (2001), for example, point out that future GHG emissions depend critically on population and productivity growth, which are uncertain. Lower growth would make global warming less of a problem. On the other hand, lower productivity growth implies a lower discount rate, which, as Pizer (1999) shows, increases the present value of the future benefits from abatement.

A number of studies have attempted to assess the sources and extent of uncertainty over the benefits from reducing GHG emissions. One of the earliest was Nordhaus (1994b), who obtained estimates of the percentage loss in gross world product from a survey of natural scientists and economists. Roughgarden and Schneider (1999) then used the Nordhaus survey results, along with other survey evidence, to back out confidence intervals for a damage function. Pizer (1999) developed an integrated climate-economy model and used it to both assess parameter uncertainty and study its policy implications. His model has nineteen parameters. Six parameters describe economic activity, and are estimated econometrically using historical data, yielding a joint distribution. Uncertainty over the remaining thirteen parameters (describing emissions growth, GHG retention and decay rates, control costs, and long-term population and productivity growth) is based on “subjective analysis.” Other studies likewise assess uncertainty using subjective analysis and expert opinion; for an overview, see Heal and Krström (2002) and Goulder and Pizer (2006).

One could argue that what really matters is the possibility and consequences of temperature increase at or above the upper end of the IPCC range. How likely is a 6°C temperature increase, and what impact would it have? One possible scenario (absent the probability estimates) was depicted by Gore (2006)—large parts of New York City would be under water. (Could New Yorkers adapt? Perhaps.) Clearly, the issue is whether, for some plausible increase in temperature, things could get really bad, to the point where adaptation could not compensate. In other words, is there a tipping point in the benefit function, and if so, where is it? Unfortunately, we do not know.

As of now, very little work has been done to assess the probabilities of catastrophic climate change, or to estimate the point (e.g., the change in mean temperature) at which a catastrophic outcome becomes likely. People have widely differing (and often strongly held) opinions about the likelihood and makings of a catastrophic outcome, but this is just symptomatic of how little we know. There have been studies of the policy implications of catastrophic outcomes, which I will discuss later, but those studies take the outcome and its characteristics as given.⁴

This problem, which I will call “tipping point uncertainty,” is not unique to global warming. For example, studies of toxic waste disposal suggest that points could be reached that are catastrophic for land and water use in localized areas. But again, there is considerable uncertainty as to where those points are.

In summary, what do we know about uncertainty over the benefits of environmental policy? As I have tried to show in the case of global warming, which has been studied quite extensively, we know very little. We know that there is a good deal of uncertainty, but we are hardly able to quantify it, especially when it comes to tipping points. We would come to a similar conclusion for environmental problems that have received less attention (e.g., acid rain and toxic waste). Given that uncertainty clearly matters (as I will explain in more detail below), we seem to have our work cut out for us.

⁴For example, Pizer (2003) modifies the DICE model developed by Nordhaus (1994a), replacing the original quadratic relationship between economic damage and temperature change with a function that is much more convex (and varies the degree of convexity). But his focus is on whether price versus quantity is the more effective policy instrument. As he says, “Unfortunately, there is little empirical information concerning either the degree of steepness or the point where the steepness begins.”

Uncertainty over Costs

For some environmental problems, particularly those with more limited time horizons, policy costs are better understood and subject to less uncertainty than are the benefits. For example, we have years of experience with limits on SO_x and NO_x emissions from coal-burning power plants. We know the cost (and effectiveness) of scrubbers and of substituting low-sulfur coal, and we can also infer costs from prices of tradable emission allowances. And while the cost function for emission controls is nonlinear (the cost of an incremental reduction in emissions rises rapidly as the emission level becomes very low), there is no “tipping point” problem.

For other environmental problems, however, cost uncertainty can be quite severe. Once again, global warming is a good example. A carbon tax may be the preferred instrument for reducing carbon dioxide (CO₂) emissions, but how large a tax would it take to reduce those emissions in the United States by, say, 20 percent?⁵ The answer depends on how responsive fossil fuel demand would be to tax-induced price changes, and this will vary from sector to sector (e.g., transportation versus residential heating). The responsiveness of fossil fuel demand in any sector to price changes in turn depends on the long-run price elasticity of energy demand in the sector, and the long-run elasticity of substitution between fossil and nonfossil energy sources. We have a reasonable understanding of energy demand elasticities, but our knowledge of the long-run elasticity of substitution between fossil and nonfossil energy sources is tenuous at best. The reason is that the ability to substitute from, say, coal to wind power for electricity production depends on the cost and availability of the latter (and that cost will partly depend on its environmental impact). Today, that substitutability is quite limited; what it would be twenty or fifty years from now largely depends on technological change, which is, again, hard to predict.

Another way to look at this is to ask whether (and how fast) the cost difference between fossil-based and alternative energy supplies will converge. Solar, wind, and biomass costs have fallen over the past twenty years, but the differences are still very large. There is reason to think that over the next fifty years, those differences may decline or even disappear as fossil fuel prices rise (because of depletion) and costs of alternative energy sources fall (because of technological change and economies of scale). Chakravorty, Roumasset, and Tse (1997) have developed an empirical model of this process that takes into account the depletion of the potentially discoverable reserves of fossil fuels. They project that the world will move towards wide-scale use of solar energy, so that “ninety percent of the world’s coal will never be used,” and global temperature will rise only about 1.5–2°C over the next century, making the cost of mitigation low indeed. However, since standard errors are missing, one cannot attach confidence intervals to these optimistic forecasts. Other forecasts suggest starkly different results, with fossil fuels remaining the predominant energy source, in the absence of major policy changes.

⁵We would also need to know how large a tax it would take to reduce emissions in other countries, and for developing countries like China and India, the uncertainties are far greater than for the OECD. Then there is the cost of free riding. How large a tax would be needed for the OECD countries if the larger developing countries did not also agree to reduce emissions?

The basic problem is that, with long time horizons, policy costs depend on technological change, which is inherently difficult to predict. In fact, it is difficult to even characterize the uncertainty over technological change. Thus, for global warming and other long-horizon environmental problems, the cost side of policy is subject to uncertainty over the uncertainty.

Implications for Policy Design

Uncertainties over benefits and costs can affect policy design in at least three fundamental ways. First, they can affect the optimal choice of *policy instrument*, that is whether pollution is best controlled through a price-based instrument (e.g., an emissions tax) or a quantity-based instrument (e.g., an emissions quota). Second, they can affect the optimal *policy intensity*, for example, the optimal size of the tax or the optimal level of abatement. Third, they can affect the optimal *timing of policy implementation*, that is whether it is best to put an emissions tax in place now or wait several years (and thereby reduce some of the uncertainty).

Choice of Policy Instrument

The implications of uncertainty for the optimal choice of policy instrument has been studied extensively, beginning with the seminal article by Weitzman (1974), who showed that in the presence of cost uncertainty, whether a price-based instrument or a quantity-based instrument is best depends on the relative slopes of the marginal benefit function and marginal cost function. If the marginal benefit function is steeply sloped but the marginal cost function is relatively flat, a quantity-based instrument (e.g., an emissions quota) is preferable: an error in the amount of emissions can be quite costly, but not so for an error in the cost of the emissions reduction. The opposite would be the case if the marginal cost function is steeply sloped and the marginal benefit function is flat. Of course, in a world of certainty, either instrument will be equally effective. If there is substantial uncertainty and the slopes of the marginal benefit and cost functions differ considerably, the choice of instrument can be crucial.

Weitzman's original result has been extended in a number of directions. For example, Stavins (1996) showed that a positive correlation between marginal benefits and marginal costs pushes the optimal choice towards a quantity instrument. Often, however, there is no need to choose exclusively between a price and a quantity instrument. A number of studies have shown that, in the presence of uncertainty, "hybrid" policies that combine both instruments generally dominate the use of a single instrument (e.g., Roberts and Spence [1976], Weitzman [1978], and in the context of climate change policy, Pizer [2002] and Jacoby and Ellerman [2004]). The optimal design of a hybrid policy depends not only on the shapes of the benefit and cost functions, but also on the nature and extent of the uncertainties. Our lack of knowledge of the uncertainties (and often the shapes of the functions) means that policy design will be suboptimal at best; much more work is needed to get a better understanding of just how suboptimal.

Policy Intensity

Uncertainties over benefits and costs can also affect the optimal policy intensity, that is the size of an emissions tax or the amount of abatement that should be mandated. If there are

no irreversibilities (which are discussed below), then in many cases uncertainty will lead to a *lower* policy intensity.

Suppose the policy instrument is an emissions quota, which we can think of as an amount of abatement between 0 and 100 percent. The greater the abatement, the greater will be the resulting benefit, and the greater will be the cost. However, in many cases we would expect the benefit function to be concave: increasing the abatement from zero to ten percent will have a large incremental benefit, whereas increasing the abatement from 90 to 100 percent is likely to have a much smaller incremental benefit. (This would be the case, for example, if a high level of pollution has serious health effects, so there is a substantial benefit even from reducing emissions by ten percent, whereas a low level of pollution has negligible health effects, so there is very little added benefit in going from a 90 percent to 100 percent emissions reduction.) Likewise, the cost of abatement is usually convex: going from no abatement to a ten percent reduction in emissions is likely to be much less costly than going from a 90 percent reduction to a 100 percent reduction. If the uncertainties over benefits and costs are proportional to the amount of abatement, it follows that the optimal amount of abatement (which equates expected benefits with expected costs) will be lower than in the absence of uncertainty.⁶ On the other hand, if there is no cost uncertainty, and the uncertainty over benefits is proportional to the level of emissions (as opposed to the abatement level), the optimal abatement would be higher than in the absence of uncertainty.

One might ask whether the possibility that little or no abatement would lead to a catastrophic outcome might increase the optimal abatement level. It is important not to confuse the *expected* benefit from abatement with the *variance* of that benefit. The possibility of a catastrophe will likely increase the expected benefit from any amount of abatement, which, other things being equal, implies a greater optimal abatement level. But if we are interested in the effect of uncertainty alone, we must keep the expected benefit fixed as we increase the variance of the benefit.

Policy Timing

Uncertainty can also affect the optimal timing of policy implementation—but only if there are sunk costs of implementing the policy, and/or the environmental damage from having no policy in place is at least partly irreversible. Depending on the particular situation, it may be optimal to defer the implementation of a policy until we learn more about benefits and costs, or to speed up the implementation to avoid irreversible damage. I will address this implication of uncertainty as I discuss irreversibilities.

Irreversibilities

It has been understood for many years that environmental damage can be irreversible, and that this can lead to a more “conservationist” policy than would be optimal otherwise.

⁶Suppose the cost of abatement is $C(A) = c[(1 + \varepsilon)A]^2$, where c is a constant, A is the percentage abatement, and ε is random and equal to either -1 or $+1$, each with probability $1/2$, so the expected value of ε is 0. Using this expected value, the cost of abatement is cA^2 . However, the expected cost of abatement is $1/2c(0 + 4A^2) = 2cA^2$. The uncertainty over ε increases the expected cost of abatement, making the optimal amount of abatement smaller.

To my knowledge, the earliest economic studies to make this point are Arrow and Fisher (1974) and Henry (1974). But, thanks to Joni Mitchell, even noneconomists know that if we “pave paradise and put up a parking lot,” paradise may be gone forever. And if the value of paradise to future generations is uncertain, the benefit from protecting it today should include an “option value,” which pushes the cost-benefit calculation towards protection.⁷ Thus, it might make sense to restrict commercial development in a wilderness area, even if *today* very few people care at all about that wilderness or have any desire to visit it. After all, if it becomes clear over the next few decades that very few people will *ever* care much about that wilderness, we will still have the option to pave it over. But if we exercise the option of paving over the wilderness today, we will lose the flexibility that the option gave us. If the social value of the wilderness rises sharply over time, the social loss from having exercised our option prematurely could be great.

There is a second kind of irreversibility, however, that works in the opposite direction: protecting paradise can impose sunk costs on society. If paradise is the wilderness area, restricting development and human access would imply a (permanently) forgone flow of wage and consumption benefits (e.g., from a large ski resort). If paradise is clean air and water, protecting it could imply discrete sunk cost investments in abatement equipment and/or in ongoing flow of sunk costs for more expensive production processes. In both cases, these costs are permanent—we cannot recapture them in the future should we decide that clean air and water are less important than we had originally thought. The point is that this kind of irreversibility would lead to policies that are *less* “conservationist” than they would be otherwise.

The implications of irreversibilities have been explored in a large and growing literature.⁸ The important questions relate to the conditions under which irreversibilities matter for policy and the extent to which they matter.

When Do Irreversibilities Matter?

It is important to stress that *these irreversibilities only matter if there is uncertainty*. To understand why, suppose that today we know *precisely* how society will value a pristine wilderness area every year over the next two hundred years, and we also know what would be the annual flow of profits, wages, and consumer surplus over that same period from the conversion of that wilderness into a large commercial resort. Finally, suppose we know the

⁷“Option value” and “quasi-option value” have been used in the environmental economics literature in different and sometimes confusing ways. Arrow and Fisher (1974) referred to the value of waiting when environmental damage is irreversible as “quasi-option value.” The term “quasi-option value” has been adopted by other authors, who use “option value” to refer to the value of delay because of risk aversion. See, for example, Conrad (1980), Freeman (1984), and Hanemann (1989). I will use the term “option value” to refer to any opportunity costs or benefits resulting from irreversibilities and uncertainty. This is consistent with the real options literature, for example, Dixit and Pindyck (1994), and with modern textbook usage, for example, Tietenberg (2006).

⁸Fisher and Hanemann (1990) and Gollier, Jullien, and Treich (2000) extend the results of Arrow and Fisher (1974) and Henry (1974), as do Scheinkman and Zariphopoulou (2001), but using a continuous-time model. Kolstad (1996b) and Ulph and Ulph (1997) address the conditions under which irreversibilities matter. Implications for the timing of policy adoption are studied in Pindyck (2000, 2002). These are some examples; for a survey, see Heal and Kriström (2002).

correct discount rates that would apply over that two-hundred-year period. We could then calculate present values and do a simple cost-benefit comparison of the pristine wilderness and the commercial resort. If the comparison favored the resort, we could allow commercial development, knowing that nothing would change in the future that would cause regret and lead to a desire to “undo” the loss of the wilderness. Likewise, if the comparison favored the wilderness, we could prevent commercial development, knowing that nothing would change that would cause regret over the loss of surplus from a resort. Irreversibility would be irrelevant.

The environmental economics literature has also been concerned with the question of when irreversibility begins to matter in terms of affecting current decisions, even if there is uncertainty (e.g., Kolstad [1996b] and Ulph and Ulph [1997]). The short answer is that irreversibility will affect current decisions if it would constrain future behavior under plausible outcomes. Consider global warming, and suppose that benefits and costs are completely linear and there is no risk aversion. Then the relevant constraint is that while we could (at great cost) have zero emissions in the future, we cannot have negative emissions.⁹ If there is a nonnegligible chance that we might want to have negative emissions in the future, the irreversibility constraint might bind, and should lead to lower emissions today. However, irreversibility could matter even if there is no chance that we would want negative emissions. Suppose, for example, that the cost of reducing emissions rises more than proportionally with the amount of reduction. In that case, we would lower emissions today by an amount of, say, x , so that we could avoid having to lower them in the future by $2x$ at a cost that is more than twice as large.

The presence of uncertainty can affect policy even if there are no irreversibilities, but in a more limited way than if irreversibilities are present. As explained earlier, uncertainty can affect policy because of nonlinear cost or benefit functions, or discount rate uncertainty. For example, suppose the environmental impact of air pollution was completely reversible so that each year society could impose a quota on the pollutant concentration, at a cost proportional to the quota. Suppose also that each year the *actual* concentration equaled the quota plus a zero-mean proportional error. Finally, suppose (realistically) that the adverse health effect of pollution rises more than proportionally with the concentration, so that the benefit loss from a five-percent unexpected increase in the concentration above the mandated quota is greater than the benefit gain from a five-percent unexpected decrease. Compared to the case of no random errors, the optimal quota would then be lower.

But now suppose that the atmospheric accumulation of this pollutant is very long-lived, so that emissions could be reduced but not the accumulation of past emissions. Because each year's actual emissions are partly random, future concentrations would likewise be random, and the variance of those concentrations will grow with the time horizon. In this case, the policy implications of uncertainty are much greater. The reason is that the range of possible future concentrations is far greater, and should those concentrations turn out to be very large, we will be stuck with them—we cannot “undo” the emissions quotas of the past which *ex post* have turned out to be too high. As a result, the optimal policy would

⁹This is not precisely correct. With near-zero emissions and a carbon sequestration program, effective emissions could be made negative. But as a practical matter, negative emissions are hard to imagine.

call for a smaller quota than would be the case if accumulations of the pollutant were very short-lived.

It is important to be clear about why irreversibility is the key. Returning to our example, the fact that accumulations of the pollutant are long-lived creates a possibility of severe regret, which is not offset by the possibility of very little pollution. A “bad-news principle” is at work here: if future concentrations of the pollutant turn out to be less than expected (“good news”), or if the economic and health impact of the pollution turns out to be less than expected (more “good news”), we could always relax the quota. But if future concentrations and/or the economic and health impact turn out to be greater than expected (“bad news”), there is little we can do to correct the situation. Even if we make our quota much more stringent, it will take many years for the concentrations to fall. It is this possibility of “bad news” that affects current policy, and the greater the uncertainty, the greater that possibility.¹⁰

In the example above, I focused on the irreversibility of environmental damage, but the optimal policy would also be affected by the irreversibility inherent in the sunk costs of abatement. To understand this, suppose that atmospheric accumulations of this pollutant were very short-lived, but for it to abate, companies would have to install long-lived equipment. Then, if we later learn that the economic and health impact of the pollutant is much less than expected, we will be stuck with the equipment—we cannot “undo” the emission quotas of the past which *ex post* have turned out to be too small. In effect, a “good news principle” would apply: it is the possibility that the harm from the pollutant will turn out to be less than expected (“good news”) that would cause regret. As a result, the optimal policy would call for a *lower* mandated abatement than would be the case if the capital equipment used to abate were very short-lived.

How Do Irreversibilities Affect Policy?

We have seen that irreversibilities can interact with uncertainty to affect current policy, sometimes making it more and sometimes less “conservationist.” But how important are irreversibilities, and what is their overall effect on policy? If we ignore them, would we be led seriously astray when designing policies to deal with, say, global warming or toxic waste disposal? And would taking them into account make us more or less “conservationist”?

As one would expect, the answers depend on the particular policy problem. Unfortunately, there are very few policy problems for which the effects of irreversibilities have been studied in any detail. Not surprisingly, the problem that has received the most attention in this context is global warming, a problem for which we have a good sense of the irreversibilities involved, and, as discussed earlier, the uncertainties are considerable. But even here we have no clear answers as to the importance of irreversibilities and the overall effect they have on policy. The reason is that to determine the optimal abatement policy for any realistic climate-economy model, one must solve (analytically or numerically) a complex stochastic dynamic programming problem. Thus, researchers have had to make strong

¹⁰This point was introduced by Bernanke (1983), and is discussed in detail in Dixit and Pindyck (1994). The extension to environmental policy (and the “good news principle” discussed below) is in Pindyck (2002).

simplifying assumptions, and different sets of simplifying assumptions have led to quite different results. This is best seen by summarizing the approaches and results of several of the studies published over the past decade.

One of the most common simplifying assumptions is to limit time to two periods, so the question becomes how much emissions should be reduced now versus in the future. This was the approach used by Kolstad (1996b) in one of the earliest studies of the opposing irreversibilities—long-lasting GHG concentrations versus long-lived abatement capital—involved with global warming. He examined how the prospect of more information about the economic impact of GHGs in the second period would affect sunk expenditures on abatement in the first period. Not surprisingly, both irreversibilities can matter, but the net effect on the first-period policy depends on the relative decay or depreciation rates of GHG concentrations and abatement capital, and on the expected benefit of abatement.

In another study, Kolstad (1996a) adapts the Nordhaus (1994a) DICE model by introducing uncertainty—which is reduced as learning occurs—over whether global warming will be a “big” or “little” problem. He calculates the optimal abatement policy over twenty ten-year periods, and finds that only the irreversibility associated with abatement capital matters, so the near-term policy should be less conservationist. The reason is that temperature change from GHG build-up is sufficiently slow so that emissions could always be slightly reduced in the future. Using a two-period model, also with functional forms and parameters taken from the DICE model, Fisher and Narain (2003) likewise find that the investment irreversibility effect is much larger than the GHG irreversibility effect, so that uncertainty over the impact of climate change leads to a reduction in first-period abatement.

In Pindyck (1998, 2000, 2002), I develop continuous-time models in which there is ongoing uncertainty over both the benefits from reduced GHG concentrations and the evolution of those concentrations, concentrations are long-lived, and there are sunk costs of policy adoption. I focus on the timing and size of a single permanent reduction in emissions (with the sunk cost of abatement a quadratic function of the size of the reduction). I find that, for a “reasonable” range of parameters, either kind of uncertainty leads to a higher threshold for policy adoption and a less stringent reduction in emissions. The reason is that policy adoption commits to a reduction in the entire trajectory of future emissions at a large sunk cost, while inaction over any short time interval only involves continued emissions over that interval.¹¹

The studies cited above, however, do not consider possible catastrophic impacts of GHG accumulations. Those that do consider catastrophic impacts find that they can lead to earlier and more stringent abatement policies, but only if the likelihood of a catastrophe is strongly linked to the GHG concentration. This ambiguous result is shown clearly in

¹¹Newell and Pizer (2003b) and Pizer (2005) also develop dynamic models to study optimal policies when GHG concentrations are long-lived (but abatement capital is not). They focus on the choice of policy instrument, and show that, despite the irreversibility, price-based policies (e.g., taxes) are strongly preferred to quantity-based policies (e.g., tradable emission allowances). Kelly and Kolstad (1999) examine the implications of learning (e.g., about parameters of the benefit function). They show that the ability to learn can imply larger emissions now, because a greater GHG concentration provides more information about parameter values. Gollier, Jullien, and Treich (2000) use a two-period model with irreversibility and learning to show how effects of uncertainty depend on the shape of the representative consumer’s utility function.

Clarke and Reed (1994), who develop a simple but revealing theoretical model in which consumption causes the emission of an accumulating pollutant (so that reducing emissions requires consuming less), and the catastrophe is a random (Poisson) arrival in which welfare is reduced permanently to zero. The hazard rate could be a constant, or it could be an increasing function of the stock of pollutant. They show that if the hazard rate is a constant, we should accept *more* pollution now, because we will all be dead (or at least have zero welfare) at some point in the future that is unrelated to how much we pollute. On the other hand, if the hazard rate is sufficiently increasing with the stock of pollutant, we should pollute *less* now, because that will lengthen our expected remaining time on the planet. If the hazard rate increases only slowly with the stock of pollutant, the impact on the current level of pollution could go either way.

In related work, Tsur and Zemel (1996) derive optimal abatement policies using a model in which an undesirable (possibly catastrophic) event is triggered when the stock of pollutant reaches a critical level—but that critical level is unknown. This kind of event uncertainty leads to a strongly cautionary behavior: the optimal emissions policy keeps the stock of pollutant within a fixed interval. Finally, Pizer (2003) studies catastrophic outcomes by replacing the quadratic relationship between economic damage and temperature change in the Nordhaus (1994a) DICE model with a more convex function, and varies the degree of convexity. But there are no sunk costs of adoption, and his focus is on whether price versus quantity is the more effective policy instrument.

So, where does this leave us in terms of the policy implications for global warming? The irreversibilities are clear, and the uncertainties are great. Should we adopt a stringent emissions reduction policy now, despite its cost, or go slowly and wait to learn more about the rate of global warming and its likely economic impact? To my knowledge, research to date does not give us the answer. Those studies cited above that ignore possible catastrophic impacts provide some evidence that we should move slowly. Those studies that do consider the possibility of catastrophic impacts suggest a more stringent emissions policy, but the catastrophic impacts in these studies are more or less assumed, rather than inferred from empirical evidence. Once again, we have a good understanding of the economic theory, but a poor understanding of its implementation in practice.

Renewable Resource Management

So far, I have characterized environmental irreversibility in terms of a stock externality—emissions cause the build-up of a pollutant (e.g., GHGs), and it takes a long time for the stock of pollutant to dissipate. But environmental irreversibilities can arise in other ways, such as species extinction or permanent loss of a wilderness area. An interesting area of environmental economics in which irreversibilities and uncertainty arise and interact in ways quite different from, say, global warming, is the management of a renewable resource, such as a fishery.

The regulation of a fishery can be a complex problem even if the resource has a single owner. The problem becomes more complex (and of greater interest to environmental economists) when the resource is common property (i.e., there is open access). In both cases, the problem is how to regulate extraction (fishing) so as to maximize the economic value of the resource. There is a large amount of literature on this topic, but most of the early

work assumed that the growth function for the resource stock is known and deterministic, so that the dynamics of the stock can be described by a simple differential equation in which the rate of change of the stock equals the growth function minus the extraction rate. Thus, left to its own, the stock of fish will increase to some natural carrying capacity, but over-fishing will lead to a stock below the optimal level, or even drive the stock permanently to zero.¹² The environmental irreversibility arises because it may take some time for the stock to recover from over-fishing, and it will never recover if it is driven to zero.

Uncertainty enters into the problem because the resource growth function is in fact stochastic, and sometimes highly stochastic. In other words, the rate of change of the resource stock is not a deterministic function of the current stock level minus the rate of extraction. Instead, the stock dynamics must be described by a stochastic differential equation. Furthermore, we often cannot observe the actual resource stock, but can only estimate its value subject to error. The optimal resource management problem then becomes a problem in stochastic dynamic programming, and its solution can be quite complicated.

How do stochastic fluctuations in resource growth affect the optimal regulated extraction rate? As with the global warming problem discussed above, there is no clear answer. Depending on the particular growth function and the characteristics of the extraction cost and resource demand functions, stochastic fluctuations could lead to a higher or lower regulated extraction rate. But there is an analogy with global warming. As a catastrophic outcome becomes a more distinct possibility, the optimal policy becomes more conservationist, that is an increase in the volatility of stochastic fluctuations in the resource stock leads to a reduction in the extraction rate. The reason is that as the stock becomes smaller there is a greater chance for a stochastic decline in the stock to (irreversibly) drive the resource to extinction.¹³

Discounting over Long Time Horizons

The role of uncertainty in policy design is especially important for environmental problems that involve long time horizons. First, it is difficult enough to predict the costs and benefits, over the next decade, of reducing air or water pollution. Over a fifty-year horizon, unpredictable technological change, changes in land and water use, and population shifts make the uncertainties over policy costs and benefits far greater. Second, discount rates are inherently uncertain, and a long time horizon makes that uncertainty extremely important. If the discount rate is “high” it will be difficult to justify almost any policy that imposes costs on society today but yields benefits only fifty to a hundred years from now, so the size of the discount rate can be the make or break factor in policy evaluation. Third, uncertainty over future discount rates impacts the choice of the discount rate that we should actually use in practice—it makes that rate *lower* than any expected future rate. I have already discussed

¹²For an excellent textbook treatment of renewable resource management and review of the early literature, see Clark (1990).

¹³I make no attempt to survey the literature on renewable resource management under uncertainty, but examples include Pindyck (1984) and, more recently, Singh, Weninger, and Doyle (2006). Weitzman (2002) shows that, even with severe stochastic fluctuations in the resource stock, landing fees dominate quotas as a policy instrument.

the first and second of these three aspects of long-horizon uncertainty, so I will focus here on the third aspect—the fact that discount rate uncertainty reduces the effective discount rate that should be used for policy evaluation.¹⁴

As explained earlier, this is a straightforward implication of the fact that the discount factor is $1/(1 + R)^T$, so with R uncertain, the *expected* discount factor is greater than the discount factor calculated using the expected value of R . A simple example might help clarify this. Suppose we want to evaluate the expected present value of a \$100 benefit to be received a hundred years from now, but we believe that the “correct” discount rate over the entire hundred years will turn out to be either zero or ten percent, each with probability 1/2. If we apply the expected value of the discount rate, that is five percent, the \$100 future benefit will have a present value of less than \$1. But in fact the expected present value of the benefit is much higher than \$1. If the true discount rate turns out to be zero, the present value would be \$100, but if the discount rate turns out to be ten percent, the present value would be close to zero. Thus the expected present value of the \$100 benefit is $1/2(\$100) + 1/2(\$0) = \$50$. Now we must ask what single discount rate when applied to a \$100 benefit received one hundred years from now would yield a present value of \$50. The answer is, about 0.7 percent. Thus even though the expected value of the discount rate is five percent, the uncertainty (it could in fact be either zero or ten percent) implies an *effective* discount rate of less than one percent.

Thus, discount rate uncertainty reduces the effective discount rate that should be used to calculate present values. Of course, the size of the reduction depends on the extent of the uncertainty. In the example I just gave, if the expected value of the discount rate was five percent but the range for the actual rate was between four and six percent, the effective rate would be only slightly less than five percent (4.58 percent, to be precise). This just means that understanding the nature and extent of discount rate uncertainty is a crucial step in doing the actual discounting, and thereby evaluating a policy that is expected to yield long-term benefits.

One approach to evaluating discount rate uncertainty is to estimate the parameters of the stochastic process followed by an appropriate interest rate. To do this, one needs a very long historical time series for the interest rate, which is problematic. Newell and Pizer (2003a) used this “reduced-form” approach by estimating an autoregressive equation for the interest rate using some two hundred years of data on government bond rates. Putting the quality of the data aside, they found uncertainty to have a substantial effect on the discount rate for horizons of one hundred years or more. For example, depending on the particular estimated interest rate equation, they find that the value today of \$100 to be received one hundred (two hundred) years from now is only \$1.98 (\$0.04) using a constant four percent discount rate, but is between \$2.61 and \$5.09 (\$0.10 and \$1.54) when uncertainty is taken into account.

Another approach is to note that the tails of the distribution for future interest rates are what really matter, and to try to estimate that part of the distribution. In the long run, changes in real interest rates will come from changes in real economic growth, so what

¹⁴By “effective discount rate,” I mean the single rate that could be used in place of the range of possible future rates. The fact that the effective discount rate is reduced by uncertainty over future rates has been discussed in detail by Weitzman (1998) and Newell and Pizer (2003a). To my knowledge, this result was first formally demonstrated by Dybvig, Ingersoll, and Ross (1996), although in a different context.

we want is the distribution for future real growth rates and, in particular, the tails of that distribution. For example, the possibility of rare disasters (not necessarily environmental in origin) would yield “fat tails” for the distribution of future real interest rates, and thereby considerably reduce the current effective discount rate. Barro (2005) extrapolates from the wars and depressions experienced by different countries over the past century, and shows that possible disasters can make the effective discount rate close to zero.¹⁵ However, the number of wars and depressions is limited, making it difficult to pinpoint the effective discount rate.¹⁶

What discount rate, then, should be used to evaluate costs and benefits over long time horizons? The studies mentioned above do not give us a clear answer. However, they do show that the correct rate should decline over the horizon and that the rate for the distant future is probably well below two percent, which is lower than the rates often used for environmental cost-benefit analysis. Thus, costly environmental policies whose benefits will come a hundred years from now may indeed be justifiable.¹⁷

Conclusions

Uncertainty is central to environmental policy. For most environmental problems, we have very limited knowledge of the underlying physical or ecological processes, the economic impacts of environmental change, and the possible technological changes that might occur and ameliorate the economic impacts and/or reduce policy costs. If costs and benefits were linear and both environmental damage and policy costs were reversible, these uncertainties would not complicate matters much; policies could be based on expected values of costs and benefits at each point in the future. But as I have tried to show, cost and benefit functions tend to be highly nonlinear, and both environmental damage and policy costs are often irreversible. As a result, it can be misleading to base policies on expected values of costs and benefits.

Then how should we base a policy? Unfortunately, there are no simple guidelines or rule of thumb for adapting environmental policies to uncertainty, at least not that I am aware of. As we have seen, the irreversibilities associated with environmental damage and policy costs

¹⁵Barro’s objective is to explain well-known asset-pricing puzzles, one of which is a near-zero real risk-free interest rate. (Another is the large observed risk premium on the market, which seems only to be consistent with an unreasonably high index of risk aversion on the part of investors.) Weitzman (2006) takes another approach to the same set of puzzles. He shows that parameter uncertainty (in terms of both the mean and the variance) with respect to the structure of the economy will lead to fat tails, which likewise imply near-zero real risk-free interest rates, and explain the large observed market risk premium.

¹⁶In an engaging book, Posner (2004) examines a wide range of possible catastrophic events ranging from flu pandemics to nuclear war to an asteroid hitting the earth. Although his estimates of the probabilities are largely subjective, the number of possible events is large enough and the economic impacts severe enough to make one think that Barro’s study is overly optimistic, and the left tail of the real growth rate distribution is quite fat indeed.

¹⁷Also, there is evidence that consumers’ subjective discount rates are much higher for short-run than long-run outcomes. For an overview of behavioral approaches to discount rates, see Frederick, Loewenstein, and O’Donoghue (2002). Weitzman (2001) also provides evidence of higher short-run than long-run rates, and shows that economists differ widely in their opinions regarding discount rates. Related discussions are in Gollier (2001, 2002).

work in opposite directions, so their net effect tends to be model-specific. Recent studies of global warming that ignore catastrophic impacts show a net effect that favors waiting over early action. This result can be turned around in models in which the health and economic impact rises at a sharply increasing rate with the amount of pollutant, for example, if at some level the pollutant could have a catastrophic impact. This would especially be the case if the point at which a catastrophic outcome would occur is unknown. However, even for well-studied problems like global warming, we know very little about the likelihood of catastrophic impacts. Furthermore, most other environmental problems (e.g., deforestation and toxic waste disposal) have received much less attention, so we know even less about the characteristics of the cost and benefit functions in these cases.

We have made greater progress in terms of understanding the discount rates that should be used to evaluate policies for which impacts occur over very long time horizons. The very fact that the marginal return on capital, and thus real interest rates, are stochastic implies that the effective discount rate should be lower than some kind of average expected rate, and a rate close to zero is not implausible. But what matters is the tail of the distribution, that is the probabilities of one or more severe economic contractions over the next century, and given our (fortunately) limited experience with severe contractions, we are left with ranges of plausible discount rates. And while zero and 2 percent may both be plausible numbers, they can have very different implications when policy benefits occur a hundred years from now.

The good news is that environmental economists have plenty of work to do, and need not plan on early retirement; likewise for physical scientists working on environmental problems. It seems to me that at least one major focus of research should be on the causes and likelihood of severe or catastrophic outcomes, and this will likely involve collaboration between economists and physical scientists. And, while global warming is an important problem, more work is needed on other problems that may eventually turn out to be even more pressing, such as the depletion or degradation of water resources, acid rain, toxic (and nontoxic) waste disposal, and the loss of wilderness and wildlife.

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Symposium:**The European Union Emissions Trading Scheme****The European Union Emissions Trading Scheme: Origins, Allocation, and Early Results**

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Introduction

Since January 1, 2005, a price has been paid for approximately half the carbon dioxide (CO₂) emissions originating from a region of the world that accounts for about 20 percent of global GDP and 17 percent of the world's energy-related CO₂ emissions. On this date, the world's first large-scale CO₂ emissions trading program, the European Union's Emissions Trading Scheme (EU ETS), began operating. Although the implementation has been uneven, as will be described more fully in the articles of this symposium, it has been effective. Despite the long time that it took for some of the twenty-five member states of the European Union (EU) to allocate emissions permits—or allowances—and to implement the electronic registries that would enable trading, a quantitative limit on CO₂ emissions was imposed and since then, a market price has been paid for CO₂ emissions by virtually all stationery, industrial, and electricity-generating installations within the EU.

This symposium, in the inaugural issue of the *Review of Environmental Economics and Policy*, focuses on key aspects of what is arguably the most significant attempt by any nation, or set of nations, to impose an effective limit on greenhouse gas emissions. The symposium is composed of three articles, each of which discusses a distinctive aspect of the EU ETS. This article is devoted mostly to the distinctive features that have emerged in the allocation of the newly limited rights to emit, called the European Union Allowances (EUAs), that are created by cap and trade programs. It also comments on the EU ETS's origins and antecedents and reports the results from the program's first year of operation. The article by Convery and Redmond (2007) titled "Market and Price Developments in the European Union Emissions Trading Scheme," focuses on the activity without which no emissions trading system would work: the market for allowances. It also describes the

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main features of the EU ETS, its institutional and legal context, and what is likely to come in future years. The article by Kruger, Oates, and Pizer (2007) titled “Decentralization in the EU Emissions Trading Scheme and Lessons for Global Policy,” discusses the unique decentralized structure of the EU ETS and its implications for both the functioning of the EU ETS and the prospects for a more global emissions trading regime.

This article is organized as follows. The next section briefly describes the origins of the EU ETS, its relation to the Kyoto Protocol, and its precedents in Europe and the United States. The following seven sections discuss various aspects of the allocation of EUAs, with a particular emphasis on the issues and problems encountered and their resolution. These include the lack of readily available installation-level data, the participants in the process, the use of projections, the choices of member states with respect to auctioning, benchmarking, and new entrant provisions, and the always difficult issue of deciding to whom the expected shortage is to be allocated. We then discuss the recently available data on emissions in 2005, what they indicate concerning overallocation, trading patterns, and abatement. The final section offers concluding remarks about the broader implications of the EU ETS, what seems to be unique about CO₂, and the role of noneconomic considerations in the allocation of allowances.

Origins and Precedents of the European Union Emissions Trading Scheme

The EU ETS would likely not have come into existence without the Kyoto Protocol, but the story of that relationship contains its share of irony. Briefly, emissions trading is an American institutional innovation in environmental regulation that was forced into the negotiations on the Kyoto Protocol by the United States in late 1997 in the face of strong opposition from the EU. Resistance to the concept continued until the new American president pulled the United States out of the Kyoto Protocol in 2001, after which European opposition to emissions trading faded. Thereafter, the EU ETS became an indispensable instrument of European climate change policy and the primary means by which the EU member states would meet their obligations under the Kyoto Protocol.

While the EU ETS is clearly motivated by the Kyoto Protocol, it is embedded in EU law in a manner that makes its implementation independent of the Kyoto Protocol. Even if Russia had not ratified the Kyoto Protocol in late 2004, thereby allowing it to enter into force, the EU ETS would have still been implemented, although perhaps with less conviction. As a further example of this relationship, the current three-year trial trading period (2005–2007) for the EU ETS is not part of any obligation under the Kyoto Protocol. However, it is aimed at assuring compliance with the Kyoto Protocol, and the second EU ETS trading period (2008–2012) coincides with the first commitment period under the Kyoto Protocol. Finally, as illustrated by the current “post-2012” discussion, the EU ETS is expected to continue beyond 2012, regardless of what happens to the Kyoto Protocol.

European and U.S. Precedents: Similarities and Differences

Although the origins of the EU ETS cannot be explained without a reference to the Kyoto Protocol, European antipathy to emissions trading was never as strong as an observer at the negotiations in Kyoto and later Conferences of the Parties might have been led to believe. As

explained more fully in the article by Convery and Redmond (2007), favorable mention of market-based instruments had been made in official documents as early as the 1990s, well before the publication in 2000 of the *Green Paper on GHG Emissions Trading* (European Commission, 2000), which launched serious consideration of CO₂ emissions trading as an integral part of climate policy in Europe. In addition, discussion and implementation of several small experiments in CO₂ trading provided further experience (Ellerman 2000). The four most important programs were the UK Emissions Trading Scheme (UK ETS), the Danish CO₂ trading program, the Dutch offset programs, and BP's internal experiment with emissions trading.¹ Although all are different in important ways from the EU ETS, they helped make emissions trading a less foreign concept. Finally, high-level commissions were established in Norway and Sweden to examine the feasibility of emissions trading, and both commissions recommended emissions trading as the primary means for meeting commitments under the Kyoto Protocol.

Emissions trading programs in the United States were closely followed by many in Europe, and comparisons are often made between EU ETS and the U.S. sulfur dioxide (SO₂) cap and trade system. While the latter did serve in many ways as a model, there are significant differences in the two programs. First, the EU ETS is much larger, covering approximately 11,500 sources, compared to about 3,000 for the U.S. SO₂ program. In addition, the level of prepolicy emissions in the EU ETS is over two billion metric tons of CO₂, versus sixteen million (short) tons of SO₂ in the U.S. program. Perhaps more significantly, the value of the allowances distributed under the EU ETS is equal to about \$41 billion (at €15/metric ton and an exchange rate of U.S.\$1.25/€1.00) versus about \$5 billion under the U.S. SO₂ program (at \$550/short ton). The only dimension in which the U.S. SO₂ program exceeds the EU ETS is in the required emission reduction—50 percent—versus the low to mid-single digits for the EU ETS. There are two other major differences between the EU ETS and the U.S. program. First, as discussed more fully in the contribution by Kruger, Pizer, and Oates (2007), the EU ETS has been implemented in a highly decentralized fashion, as might be expected of a multinational system. This is in stark contrast to the highly centralized implementation of the U.S. SO₂ program, which is under one sovereign jurisdiction. The second major difference is that one program limits CO₂ emissions and the other a more conventional pollutant that had been long controlled before the emissions trading program started. Although one type of emission might seem like another, at a more practical and

¹The UK ETS, which started in 2002, is a mixed system that combines installations that accepted a cap in return for an incentive payment with a baseline-and-credit system intended to introduce flexibility into the Climate Change Agreements that had been negotiated between industry and the government. The Danish CO₂ trading system, which started in 1999, was limited to the electricity utility sector. It included a safety valve feature at a relatively low level (approx U.S.\$7/ton CO₂). A “safety valve” approach ensures that costs do not become excessive by capping the cost per ton of CO₂ at a specified maximum level. The Dutch programs were solicitations for Joint Implementation (JI) or Clean Development Mechanism (CDM) project credits intended to promote the cost-effective achievement of the Netherlands' Kyoto obligations. BP's emissions trading program imposed a greenhouse gas emissions limit on operating entities in many parts of the world, but it was voluntary and internal to the corporation (Victor and House, 2006).

political level the two are viewed as being very different with many implications, not the least of which concerns the allocation of allowances, the subject to which we now turn.²

Before engaging in the discussion of allocation issues in the EU ETS, we want to emphasize that the EU ETS is pioneering in its allocation of CO₂ allowances to individual emitters.³ The Kyoto Protocol and the European Burden Sharing Agreement (BSA) allocated rights to emit greenhouse gases to nations, but not to individual legal entities. Earlier cap and trade programs in the United States have distributed allowances for conventional pollutants, such as SO₂, but not CO₂.

The Problem of Installation-level Data

Without a doubt, and to the surprise of many, the biggest problem in allocating allowances in the EU ETS was the absence of readily available installation-level data.⁴ Consequently, the collection and reconstruction of these data absorbed a large amount of resources and attention in what was already a very ambitious schedule for implementing the EU ETS.

Sources of the Data Problem

The problem was created first by the Emissions Trading Directive's requirement to allocate at least 95 percent of EUAs to emitting installations and second by the fact that the CO₂ inventory data, which had been collected and reported under the UN Framework Convention on Climate Change (UNFCCC), were based on fuel-use aggregates that did not break out emissions at the installation level.⁵ Even countries such as Germany and Sweden, that thought they had collected relatively good installation data for other purposes, found that they contained unacceptable errors. The only country for which this data problem did not exist was Denmark, which was already collecting emissions data at the installation level as part of its earlier CO₂ trading system.

The data problem was compounded by the low threshold —20 MW thermal rating—for the inclusion of combustion sources. While this low threshold minimized competitive

²The remainder of this article draws heavily from the concluding chapter in the forthcoming *Allocation in the European Emissions Trading System: Rights, Rents, and Fairness* (A. Denny Ellerman, Barbara Buchner and Carlo Carraro, eds.) to be published by the Cambridge University Press. Other general sources for information and analysis of allocation in the EU ETS are Betz, Eichhammer, and Schleich (2004), Öko-Institut (2005), DEHSt (2005), Zetterberg et al. (2005), and Dufour and Leseur (2006).

³The earlier Danish CO₂ trading program is the only exception, although the allocations were generally nonbinding and the effective instrument was a tax that was activated mostly when Danish coal-fired generators exported power to the rest of Scandinavia. The UK ETS did not involve allocation as usually defined since allowances were distributed to the winning bidders for incentive payments to join the scheme by accepting a cap on their emissions and a like endowment of tradable allowances.

⁴This problem was noted in an early study on allocation alternatives (Harrison and Radov, 2002): "No single EU database currently provides plant-level information that could be used as a solid foundation for plant-level allocations across the member states."

⁵The Emission Trading Directive, which was agreed to in late 2003, provides the legal authority for the EU ETS as explained more completely in Convery and Redmond (2007). The UN Framework Convention on Climate Change, negotiated in 1992 in Rio de Janeiro, provides the international structure for climate change policy and requires signatories to report greenhouse gas emissions, among other things.

distortions among installations in sectors included in the EU ETS, it greatly complicated the data problem, since typically, 80 percent or more of the sources accounted for 10 percent or less of emissions, and the data problems were proportional to installations and not to emissions.

Consequences of Data Problems

These initial data problems resulted in two consequences. First, the data had to be obtained directly from the sources being regulated through a voluntary effort because there was no existing legal authority to collect the data within the time frames required for implementing the system. However, industry was cooperative (perhaps because the allocations to installations depended on these data), and participants in the process reported relatively few cases of fraudulent submissions.⁶

The second consequence of the data problem was that some options for allocating allowances that might have been preferred were simply impractical. For instance, basing allocations on the 1990 emissions in order to reflect “early action,” and to be consistent with the Kyoto Protocol base year, was not feasible.

In contrast to some of the other allocation issues in the EU ETS, lack of data was a one-time problem that has now largely been overcome. The necessary data have been collected, allowances allocated to installations, and emissions data are being reported on an installation-by-installation basis. The important lesson for any future emission trading programs is that if allowances are to be allocated to incumbents (i.e., any installation covered by the EU ETS that was in operation when allowances were allocated), as has been the general pattern to date, installation-level data must be available.

Participants in the Allocation Process

The main participants in the allocation process were the European Commission, the member state governments, and the industrial firms that were to be included in the scheme and would be the main recipients of allowances. The role of these participants varied according to the two main issues to be decided: the “macro” decision concerning the total number of allowances to be created by each member state, and the “micro” decision concerning how this total would be allocated to affected firms in each member state. Each member state took the initiative in proposing in its National Allocation Plan (NAP) a total and in specifying the allocation to installations, but both aspects were subject to review by the commission. As discussed in the article by Kruger, Pizer, and Oates (2007), the decentralized, parallel processes by which these decisions were made created some problems. Here, we focus on the role of the commission in coordinating these processes and on the participants in the micro level decisions.

⁶For example, the German Emissions Trading Authority found few serious discrepancies when it followed up with a later collection of verified baseline emissions data for installations (private communication from Martin Cames). A number of the contributions in Ellerman, Buchner, and Carraro (2007) reported similar findings.

Role of the European Commission

As a coordinating agent for the whole, the European Commission contributed in many ways to the success of the EU ETS, but its most important roles were to enforce scarcity and to ensure trading. The commission enforced scarcity through its review of the total number of allowances proposed by member states and reducing this figure as required. The criterion for approval was that the total proposed by the member state be the lesser of either Business-As-Usual (BAU) emissions, or a level that would not preclude achievement of the member state's Kyoto or BSA obligation in the 2008–2012 period. Although neither of these criteria could be precisely defined, they did provide the basis for reducing the proposed totals in fourteen of the twenty-five proposed NAPs by an annual amount of slightly less than a hundred million tons of CO₂, or about 5 percent of the EU cap.

The commission ensured trading by rejecting provisions in the proposed NAPs that would have allowed an “ex post adjustment” of allowance allocations based on performance during the compliance period. It would have been more in keeping with past practices to allow quotas to be reassigned ex post, but it would also have diminished the cost imposed on CO₂ emissions and substituted a political process for reallocations of quota that the market would otherwise perform. The commission's stance on this subject marks a sharp break from the traditional application of environmental regulation in Europe, where flexibility in implementation is provided through negotiated arrangements between affected firms and the government instead of by trading allowances.

The European Commission undertook an important additional role as both educator about emissions trading and facilitator of member state decisions. This role was exercised through the initial *Green Paper on GHG Emissions Trading* (European Commission 2000),⁷ subsequent guidance documents and research reports to familiarize participants with various aspects of emissions trading,⁸ the formation of Working Group 3 to provide a forum for the exchange of information about member states' experience in allocating allowances, and numerous informal and back-channel communications with the relevant individuals in the member states. In all, the commission's exercise of this role showed a technical competence and political astuteness that contributed significantly to the success of the EU ETS.

Participants at the Member-state Level

In contrast to its prominent role in deciding member-state totals, the commission was not involved in the allocations of allowances to firms within member states, despite the not infrequent calls for greater harmonization of these distributive provisions. The main participants in the micro aspects of allowance allocation were the member state governments, usually the environmental ministry with the close collaboration of the economics or trade ministry, and the firms whose installations were covered by the EU ETS. The required public comment period was open to all, but generally speaking, other groups, whether consumer

⁷This communication provided the first detailed discussion of the possibility of establishing a tradable-permit system, explained how such a system might operate, and thereby launched the debate within Europe on the suitability of such a scheme.

⁸Examples are European Commission, DG Environment (2003), Harrison and Radov (2002), and PriceWaterhouseCoopers (2003).

representatives, environmental NGOs, or nonaffected industries, were absent or without significant influence on the final outcome. There was simply too much to do in too little time to allow for lengthy discussions involving more than the key players. The process by which allowances were allocated can be characterized as repeated iteration between government and industry, in which data were obtained, verified, and refined, while various principles of allocation were applied, until a solution which the government and the affected firms could accept was reached. The government's role was not so much to advocate any particular principle of distribution as it was to facilitate a process to evaluate and resolve competing claims and to be the final arbiter in cases where agreements could not be reached among the affected firms.

The Role of Projections

Emissions projections have always been necessary to estimate the costs of emissions trading programs, but they had not previously played as large a role in allocation as they do in the EU ETS. There are several reasons for this. First, the commission's guidance on allocation (European Commission, DG Environment 2003) explicitly suggested that projected emissions or activity levels might be a basis for allocation. Second, one part of the criterion for assessing country emissions totals—a total no greater than BAU emissions—effectively required a projection. In practice, even those member states subject to the alternative criterion (not precluding achievement of the country's Kyoto/BSA obligation) set caps that were only slightly below BAU emissions. Finally, the decision taken by many member states to endow the nonpower sectors with as many allowances as they were expected to “need” (as discussed below) implied a projection of BAU emissions for the nonelectricity sectors.

The reliance on emissions projections also suffered from its own form of data problem. Although there were a number of models that predicted national emissions, none captured the trading sector as defined in the Emissions Trading Directive.⁹ Moreover, there were few sector-specific emission prediction models. Consequently, allocation often had to wait for improvements in modeling, which depended in turn on the availability of the appropriate data. These problems were overcome, but in the end, the models used were not as fully validated and reliable as might have been desired.

While reliance on projections opened up all the problems of trying to predict an inherently uncertain future, their usage also had some benefits. Particularly in Eastern Europe, but also for allocations to nonelectricity industrial sectors, projections forced a top-down discipline at the member-state level on what were typically expansive bottom-up estimates of need from firms. They also tended to focus the debate about aggregate quantities at the national and sector levels on factors that could be evaluated empirically, such as expected rates of growth in economic activity, trends in emissions reduction, or the effects of atypical weather and other regulatory provisions. These factors were, in fact, the primary means by which the

⁹Annex I of the Directive lists the categories of activities covered by the EU ETS (European Commission 2003). Currently, energy activities (electric power, oil refineries, coke ovens), production and processing of ferrous metals (metal ore and steel), mineral industry (cement kilns, glass, ceramics), and industrial plants that produce paper and pulp are included.

European Commission challenged the overly generous emission totals proposed by many member states.

The Role of Auctioning

Perhaps no aspect of allowance allocation is of more interest to economists—and more advocated by them—than auctioning.¹⁰ Although the choices made in the first period allocations have been a disappointment to advocates, there is a clear intent to increase the share of auctioning in subsequent trading periods and some signs that this intent will be realized. Both the disappointment and the intent are captured in the EU Emissions Trading Directive. The Directive allows member states to auction up to 5 percent of their allowance total in the first trading period and up to 10 percent in the second period. Thereafter, the amount is unspecified, although it is generally expected to be greater than 10 percent. This captures the intent. The disappointment is that this language requires that at the least either 95 or 90 percent be allocated for free and that a 100 percent free allocation is not precluded.

In the development of the first period NAPs, nearly all governments explicitly raised the auctioning option as a possible design feature in the required public comment period, but for most, that was as far as it went. The nearly uniform response from the firms covered by the EU ETS was opposition. Whether for this reason or others, such as the pressing time deadlines, dependence on firms for data, or the requirement to allocate at least 95 percent for free, only four governments chose to exercise the auctioning option: Denmark, Hungary, Lithuania, and Ireland, for 5, 2.5, 1.5, and 0.75 percent of their respective totals. For the EU ETS as a whole, with an average annual allocation of almost 2.2 billion EUAs, slightly less than three million EUAs—or 0.13 percent—are designated for auctioning. In addition, a small number of EUAs may be auctioned in 2007 by countries disposing of any allowances remaining in their new entrant reserves.

In allowing the share of auctioned allowances to increase to 10 percent in the second trading period, the Emission Trading Directive signals the intent to rely more on auctioning in the future; however, the relevant question is the extent to which this indicative guidance is followed. The NAP process for the 2008–2012 period is not complete; however, of the nineteen NAPs that have been submitted to the commission for approval as of late October 2006, seven propose to auction from 0.3 percent (Belgium) to 7 percent (UK) of their totals. Although none of these plans has been approved by the commission, these seven countries propose to auction 3.4 percent of their totals, which would be 1.1 percent of the totals submitted to the commission so far. While still well below 10 percent, both the number of countries proposing some auctioning and the totals to be auctioned are increasing.

The decision to distribute nearly all the EUAs for free did lead to a significant outcry about “windfall profits,” especially from energy-intensive industrial firms, when electricity prices increased in the course of 2005 as a result of both higher energy prices and the new carbon price. This criticism tended to ignore the extent to which the higher electricity prices could be attributed to carbon prices instead of higher fuel prices, and whether the higher profits reported by electricity utilities were due to their allowance holdings or having cheap

¹⁰For a typical example in the context of the EU ETS, see Hepburn et al. (2006).

nuclear or coal generation in dispatch areas where the marginal price was set by natural gas. In any case, regulators initiated formal investigations into windfall profits in several countries, including the Netherlands and Germany.

A Shortage of Allowances

A binding constraint on CO₂ emissions implies that there will be a shortage of emissions allowances. Broadly speaking, this expected shortage for the EU as a whole was allocated to the EU15 and, within the EU15, to the electricity utility sector.¹¹ As will be discussed more fully later, the 2005 data confirm this allocation of the expected shortage.

Allocations at the Country Level

At the member-state level, the allocation of the shortage to the EU15 resulted from the structure of the member-state commitments under the Kyoto Protocol and the European BSA. Because the new East European member states do not have a problem with meeting their Kyoto obligations, the only issue under the commission's criteria for approving NAPs was whether a country's allowance total was a reasonable estimate of expected BAU emissions. For the EU15, about half of the countries are not expected to be able to meet their Kyoto obligations without additional measures. Consequently, approval of their NAPs depended on a demonstration that the proposed allocation to the trading sectors would not preclude achievement of the Kyoto/BSA target when planned government purchases of JI or CDM credits and other measures in the nontrading sectors of the economy (i.e., sectors that are not covered by the EU ETS) were taken into account. Typically, this meant that the allocation to the trading sectors would be less than the expected BAU emissions of those sectors, or that the overall EU ETS shortage would be allocated to those EU15 member states that face problems in meeting their Kyoto/BSA targets.

Allocations at the Sector Level

Most EU15 countries were very explicit about how they would allocate the expected shortage of EUAs: the shortage was to be assigned to the electricity utility sector.¹² The rationale was, first, electricity utilities did not face nonEU competition and would therefore be able to pass on the added cost of abatement to consumers. In contrast, industrial firms were competing in world markets, which would not allow them to pass on any added costs. A second rationale for allocating the shortage to the power sector was that abatement would be cheaper for electricity utilities so that they would need fewer allowances.

¹¹The EU15 comprises, in the order of their accession to the EU, France, Germany, Italy, Belgium, Luxembourg, the Netherlands, the United Kingdom, Ireland, Denmark, Greece, Spain, Portugal, Austria, Finland, and Sweden. On May 1, 2004 the EU membership rose to twenty-five with the accession of Cyprus, the Czech Republic, Estonia, Hungary, Latvia, Lithuania, Malta, Poland, Slovakia and Slovenia (referred to as EU10). On January 1, 2007, Romania and Bulgaria also acceded, thus making the total twenty-seven.

¹²A few countries such as Germany made a distinction between combustion and process emissions and treated the latter more liberally.

In practice, the process of allocation at the sector level was a matter of projecting industrial sector emissions, setting aside some allowances for various reserves, and allocating what remained to the power sector. In most cases, what remained for the power sector was less than their expected emissions before any abatement actions.

New Entrant and Closure Provisions

Two distinctive features of allocation in the EU ETS are the provisions for free allowances to new entrants and the accompanying requirement that allowance endowments be forfeited for any facility that shuts down.¹³ These features are not found in other cap and trade systems, where, with few exceptions, new entrants must purchase whatever allowances they may need and the owners of facilities that cease to function are allowed to keep their initial allowance endowments.

The explanation typically given for the new entrant provisions is a desire to avoid putting member states at a disadvantage in the competition for new investments. The corresponding explanation for the closure provision is to avoid providing an incentive for shutting down and relocating existing production facilities. In addition to these employment concerns, comments are often heard that it does not seem fair to award allowances to incumbents while denying them to new entrants, or to continue providing allowances to facilities that no longer need them.

Although the distortional effects of these provisions—either by subsidizing production or biasing technology choices in a more CO₂-emitting manner—were well known to officials in the member states and at the European Commission, they were unable to resist the political demands for the provisions. These demands did not come from incumbents who overwhelmingly favored retention of allowances upon closure and who, by definition, did not represent new entrants. Typically, they came from the highest levels of government, presumably reflecting policy concerns about effects on industrial activity and employment. In Ireland, for example, where the allocation process was conducted with more attention to economic considerations and was relatively insulated from lobbying, one of the few changes made by the cabinet in its final review and approval was to override the recommendation not to provide free allowances to new entrants, as well as that to permit allowances to be retained upon closure of a facility.

Details of the New Entrant Provisions

The number of allowances set aside for new entrants reserved by the 25 member states for the first three years of the EU ETS is 195 million tons, or about 3 percent of the total. The percentage set aside varies greatly, country to country, from as little as 0.4 percent in Poland to 26 percent in Malta. Distribution is generally on a “first-come-first-served” basis, but what countries would do if the reserve is exhausted varies. For most countries, latecomers will have to resort to the market. However, Italy and Germany have stated that the government will purchase allowances on the market to provide for all new entrants.

¹³For analyses of these provisions see Åhman et al. (2005) and Ellerman (2006).

Provisions also differ if the new entrant reserve is not fully used. Sixteen of the twenty-five member states will sell unclaimed EUAs in 2007. Six countries, including Germany, France and Spain, who have 22 percent of the total new entrant reserves, have said they will annul any remaining new entrant reserves. The remaining three member states, Cyprus, Slovakia, and Malta, have not indicated what they will do with their unused reserves.

Finally, the criterion for determining the number of allowances that can be awarded to new entrants differs considerably among member states. All of them indicate a desire to allocate allowances according to a benchmark that is based on some definition of best practice or technology multiplied by expected production or new capacity. However, the exact definition is usually vague and does not explicitly exclude expected emissions. Moreover, the indicated benchmarks can differ by fuel or technology used, especially in the electricity sector. For instance, the United Kingdom, Denmark, and Spain use a common new-entrant benchmark for all fuels, while most other countries, notably Germany and Italy, differentiate by fuel. Sweden is unusual in limiting new entrant endowments to industrial and district-heating facilities, implicitly excluding fossil-fuel-fired generating units (of which none are planned to be built).

Closure Provisions

Closure provisions also vary from country to country. Sweden and the Netherlands are unusual in allowing closed facilities to retain allowances at least until the end of the trading period.¹⁴ The most interesting variant is the “transfer rule” that was pioneered by Germany and adopted by some other member states. It allows the owner of a closed facility to transfer unused allowances to a new facility which is thereby not eligible for allowances from the new entrant reserve. These transfer rules can be very complex in defining how many allowances can be transferred, the ownership of the closed and the new facilities, and how much time can elapse between the closure of one facility and the opening of the new facility. However, one general rule is found everywhere: the transfer must be to a new facility situated within the same member state. In Germany, where concerns about production moving away to Eastern Europe are high, the transfer rule provides some incentive to maintain production at home.

The Use of Benchmarking

Benchmarking is a principle of allocation whereby some index of historical activity or capacity is multiplied by a usually uniform emission-rate standard to determine allocations to individual installations.¹⁵ Despite strong advocacy in its favor, benchmarking has not been the principle of general allocation in the EU ETS. Instead, allowances have been allocated to individual installations according to their shares of emission within the sector.

¹⁴The expectation is that closed facilities would not appear on the installation list in the NAPs that would allocate EUAs in subsequent periods.

¹⁵The idea behind benchmarking is that two installations that are alike in all respects except their levels of emissions be treated alike. A facility with higher emissions would not be allocated more allowances than an identical facility with lower emissions.

This is a distinctly different principle from what has been applied in the U.S. cap and trade programs where (although the term is not used) allocation follows a benchmarking principle using historical production shares.

The primary reason for allocating allowances according to emission shares, and not by benchmarking, is the heterogeneity of production processes and conditions at existing installations. An allocation embodying some benchmark was tried in many member states, but was not adopted because the disparities between the resulting installation-level allocations and recent or expected emissions were too great to be acceptable to the parties involved. The problem was that the installations were less alike than benchmarking assumes. Just as no one would propose that the benchmark for the steel industry should be the same as that for cement or electricity, so it was found that firms within various industry categories for which a benchmark was proposed were not homogeneous. For example, the level of emissions associated with producing steel from a blast furnace operation is different from that of an electric arc furnace, and no single steel industry benchmark could treat both alike. Every industry has similar sub-processes and products that would have to be taken into account to develop a set of benchmarks that would be perceived as fair and acceptable. The Netherlands, for example, made a serious attempt at developing a benchmarked allocation, but abandoned it after 125 benchmarks were developed. This heterogeneity was true even for electricity, where the product is homogeneous but the conditions under which it is produced are not. When the German electricity utility industry was asked to present a proposal for an appropriate benchmark, it proposed thirty-six different standards. The problem faced in the allocation process is reflected in a survey conducted for the commission concerning stakeholder views on the EU ETS. More than three-quarters of industry participants indicated that they favored benchmarking, but only if there were sufficient benchmarks to allow for specific situations within their industry (European Commission, DG Environment, 2005).

There were additional factors working against the adoption of benchmarks. Pressing deadlines and data that were not always readily available favored the simplest solution: emissions. Even if the appropriate output or input data had been available, there was not enough time to agree upon the necessary industry and sub-industry classifications for the benchmarks. Another factor complicating the use of benchmarks was the lack of any preexisting standard with legal or institutional precedent and force. In the case of the U.S. SO₂ program, where a benchmark was adopted despite significant heterogeneity in emission rates, the standard or benchmark was the New Source Performance Standard (NSPS) of 1.2 pounds of SO₂ per million Btu that was adopted following the enactment of the Clean Air Act Amendments of 1970. Approximately 60 percent of the electricity produced by generating units conformed to this standard (or a later one) when the Acid Rain Program was enacted in 1990. Nothing comparable in institutional presence and legal force exists for CO₂.

While benchmarking was generally not chosen, there were some exceptions, the most notable of which was for new entrants who, by definition, have no historical emissions. In addition, in Spain, several industries received benchmarked allocations where the benchmark was the average sector emissions rate per unit of capacity. This benchmark was chosen because reliable firm-level data on emissions and output could not be obtained. Spain also applied benchmarks to existing facilities in the electricity utility sector that were differentiated

by fuel and with some allowance for deviations from the fuel-specific benchmark for existing coal-fired plants. Italy and Poland allowed some sectors to choose between firm-level allocations based on recent emissions, projected emissions, or a benchmark. A few of these sectors chose benchmarks. Finally, Denmark applied a very specific benchmark to existing facilities—0.56 EUAs per MWh—which was derived by dividing the EUAs remaining after providing for the nonelectricity sectors by average annual generation during the baseline period of 1998–2002. Heterogeneity was not an issue here because the two Danish power generators have similar generating profiles.

Aside from these exceptions, the predominant basis for allocation to installations was the installation's share of recent sector emissions, defined by some baseline period that sometimes extended back as far as six years. In the end, the easiest data to obtain were emissions and, in the absence of an alternative principle to which most could agree, historical emissions shares became the least controversial basis for allocation.

A First Look at the 2005 Emissions Data

The release of the 2005 emissions data in April and May 2006 revealed that the number of allowances distributed to installations in 2005 exceeded those installations' emissions by about eighty million tons or about 4 percent of the total EU cap. This information caused the price for first period allowances to fall by half and the second period futures price to fall by a third.

Overallocation Becomes an Issue

The revelation of this overall long position and the associated plunge in EUA prices also changed the nature of the debate about the EU ETS: "overallocation" has taken the place of high EUA prices as the main topic of concern. This concept is not well defined and it tends to be conflated with being long; but it is generally taken to mean that too many allowances were handed out and even meant to imply that the cap is nonbinding and that there has been no reduction in CO₂ emissions.

Among other things, these data confirmed the allocation of the expected shortage of allowances to the power sector. Figures 1 and 2 aggregate installation-level data to indicate long (more allowances than emissions) and short (fewer allowances than emissions) positions by member states (Figure 1) and by EU-wide sectors (Figure 2). These positions are expressed as a percentage of the 2005 allocation to installations for the constituent member states or sectors. Long positions are shown to the right of the axis and short positions to the left with the darker shade indicating the net position. Figure 1 shows that all the countries with a net "short" position (i.e., the UK, Ireland, Spain, Italy, Austria, and Greece) are EU15 member states. Although the "long" countries are almost evenly split among the EU15 and the new EU10, all the latter are long and often by a significant percentage.

Figure 2 provides the same data by sectors across the EU. It shows that the electricity sector as a whole is short, while the nonelectricity industrial sectors are all long, in many instances, by a large percentage.

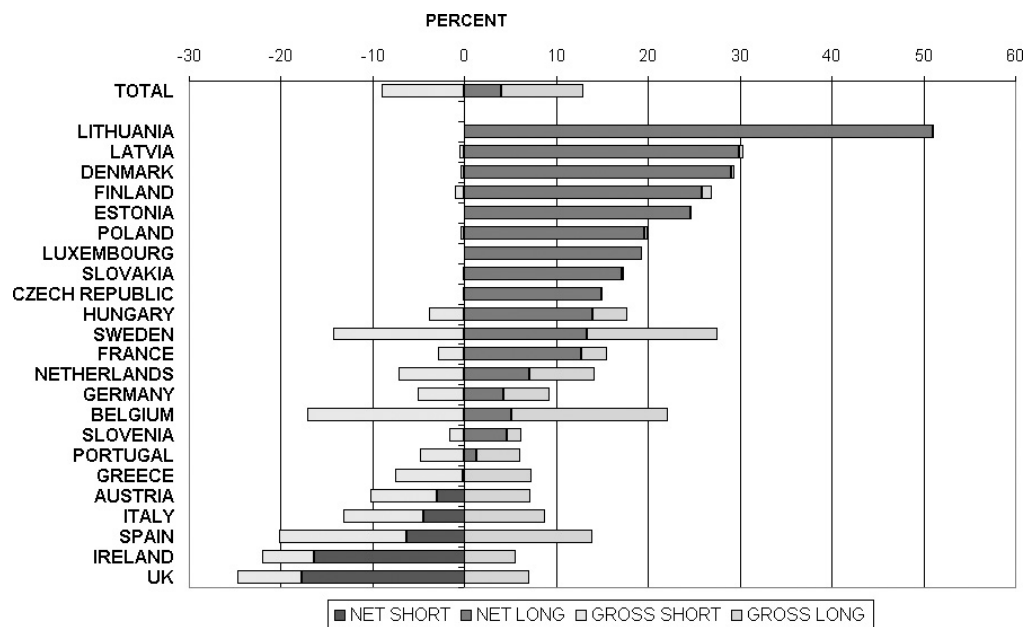


Figure 1. Short and long positions by member state.

Source: Community Independent Transaction Log (2006) and Kettner et al. (2006).

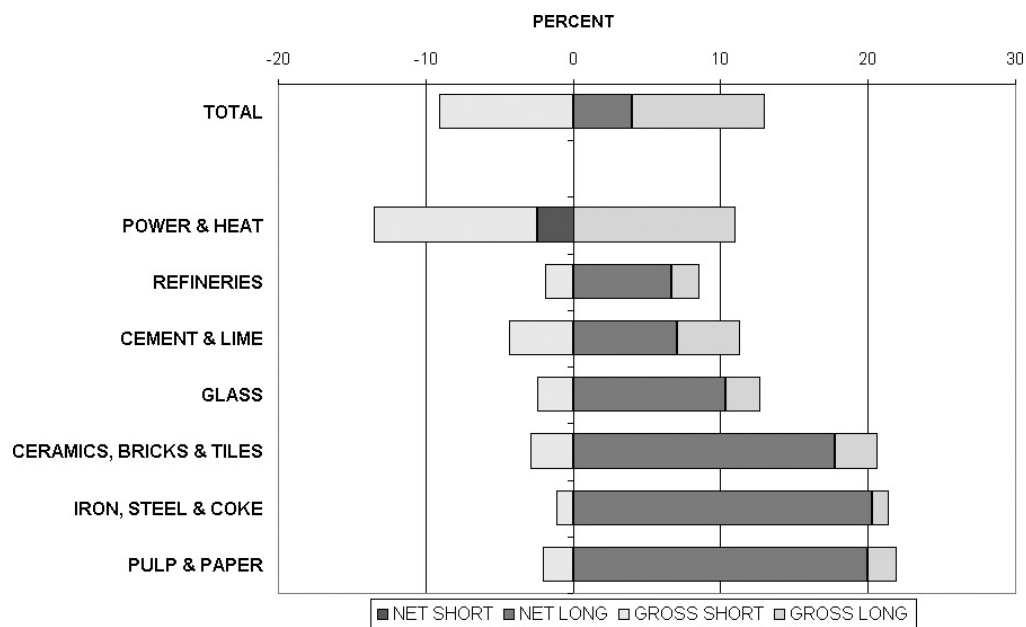


Figure 2. Short and long positions By EU-Wide sectors.

Source: Community Independent Transaction Log (2006) and Kettner et al. (2006).

While these observed short and long positions could also reflect changes from expectation¹⁶ and abatement, they confirm what would be expected based on the criteria adopted by the commission and member states in allocating the expected shortage. Although never expressed as such, the application of the criteria for approval of member state totals effectively allocated the shortage to the EU15. Also, among the EU15, the general practice was to allocate expected shortages to the power sector.

The concept of overallocation lacks a precise definition and it depends somewhat on the eye of the beholder. For instance, do the allocations to the industrial sector constitute overallocation? The directional result revealed by the 2005 data was fully intended in view of the competitive position and abatement potential perceived of these installations. But was the extent revealed by the data also intended? Interested readers can refer to Ellerman and Buchner (2006) for a more complete discussion of how overallocation might be defined more rigorously, but the general idea is that in a cap and trade system like the EU ETS, one could expect that differing abatement possibilities and economic circumstances would cause installations to be both long and short and for the net balance to be a relatively small percentage of the total allocation. From this standpoint, and referring again to Figure 1, the EU as a whole is relatively balanced, with short installations in need of additional allowances equal to about 9 percent of the total EU25 allocation, and long installations holding excess allowances (i.e., not needed to cover 2005 emissions) in an amount equal to about 13 percent of the EU cap. Several member states are, however, unbalanced in that virtually all installations were long and the net long position is large (perhaps greater than 15 percent) in relation to the member states' total. While individual circumstances could possibly explain some of these instances, such large and unbalanced long positions create a strong presumption of overallocation, mostly in the East European member states and in the nonpower sectors.

Indicated Patterns of Supply and Demand

The same data also reveal the sources of potential demand and supply of allowances in the EUA market. Whatever the reason for being short or long—whether it be higher or lower output, a more or less generous allocation, or even abatement—installations that are long are potential sellers and those that are short, potential buyers. Displaying the same data in absolute terms, as shown in Figures 3 and 4, provides a better sense of the potential trade flows.

The most striking result is that most of the market for EUAs, on both the buying and selling sides, originates in the power sector. That sector, which accounts for more than 60 percent of emissions in the EU ETS, constitutes about 90 percent of the potential demand and 50 percent of the potential supply. Looking now at the breakout by member states, the main point to note is that those member states comprising a large percentage of the

¹⁶For instance, Danish emissions vary greatly, depending on demand in the rest of Scandinavia for Denmark's large coal-fired generating capacity. Demand for Denmark's coal-fired capacity is high when hydroelectric conditions in Sweden and Norway are poor. Because hydroelectric conditions were very good in Scandinavia in 2005, the demand for Danish power was considerably below the long-run average upon which the allowance totals were calculated. The same circumstance and an extended strike in the pulp industry explain most of the Finnish surplus.

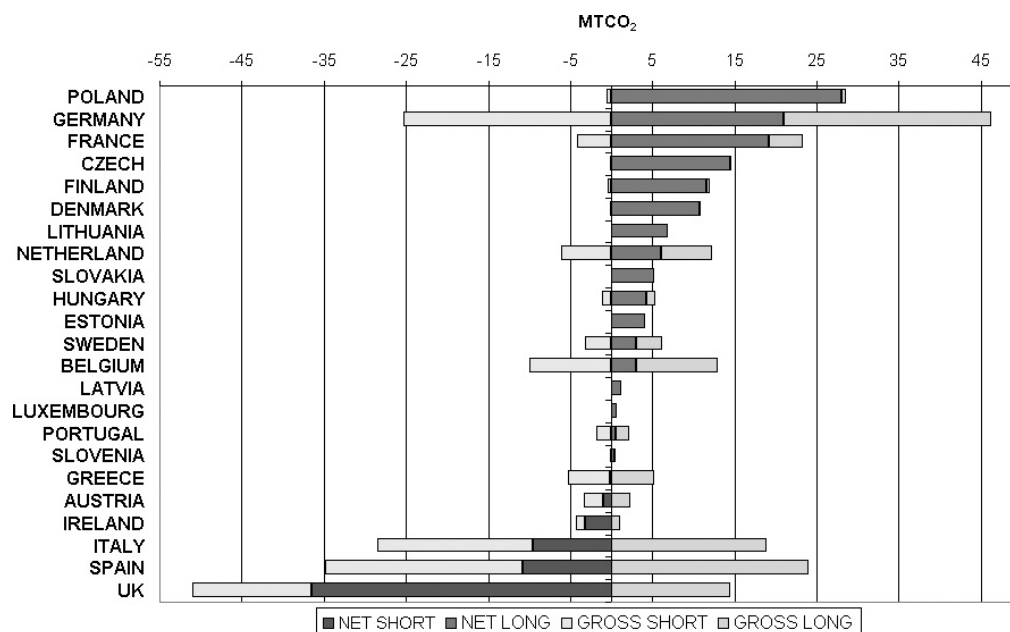


Figure 3. Short and long positions by EU member states in absolute terms.

Source: Community Independent Transaction Log (2006) and Kettner et al. (2006). The total for the EU23 is not included (as in Figure 1) because the required scale would make the comparisons among member states less evident. as of October 31, 2006. The tally for the EU23 was a gross short position of 180 million tons, a gross long position of 260 million tons, and a net long position of eighty million tones.

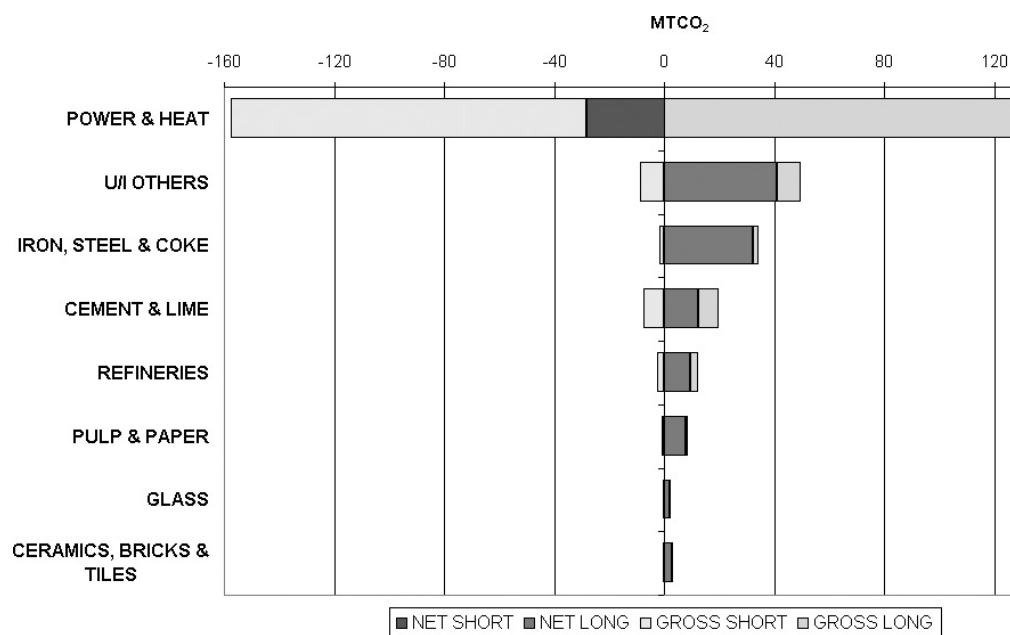


Figure 4. Short and long positions by EU-Wide sectors in absolute terms.

Source: Community Independent Transaction Log (2006) and Kettner et al. (2006).

potential demand for EUAs (the UK, Spain, Italy and Germany), are also significant sources of potential supply. Although the data on actual trades are not published, these patterns imply that a significant part of the market consists of trades among installations within the same member state. Transfers among member states are indicated by the net positions, denoted by the bold colors. Nearly all the demand for transfers among member states originated from the United Kingdom, Spain, and Italy, while the potential supply was more broadly distributed with Poland, Germany, and France being the largest potential sources.

These data and the inferences that can be drawn from them about trading patterns must be used with caution because it is also possible for installations to cover short positions in 2005 by borrowing from the installation's 2006 allowances. A recently released analysis by the Italian registry operator (Point Carbon 2006) indicates the relative importance of different means of covering a short position. This analysis reveals that of the 43.4 million tons by which Italian installations were short in 2005, 7.9 million tons were covered by borrowing from the 2006 allocation and seven million by purchases from outside Italy. Thus, 66 percent of the gross short position was covered by redistribution of allowances among Italian installations, 18 percent by borrowing from the installation's forward allocation, and 16 percent by purchases from abroad. Italy's net short position of 9.2 million tons implies that the Italian installations that were long in 2005 held a total of 34.2 million EUAs. Since only 28.5 million of these EUAs (or 83 percent) were used to cover emissions at installations that were short, the remaining 5.7 million (or 17 percent) were either carried over for use in 2006 or sold to installations outside Italy.

Abatement

Emissions data are also of interest for comparison with earlier historical data and with estimates of what emissions would have been in the year if the trading program had not been in effect. This comparison indicates the extent of emissions reduction, which is the ultimate environmental goal of an emissions trading program. Making this comparison for the EU ETS is greatly complicated by two factors: the absence of good historical data for the installations included in the program, and the effect of uncertainty on estimates of abatement when abatement goals are modest.

As noted earlier in this article, emissions data concerning the installations included in the EU ETS were not available prior to the start of the program. The only data to which reference can now be made are what were collected in the allocation process. Those data have the merit of reflecting recent emissions, but they were collected in a hurry and they are suspected of being biased upward since allowance allocations would be influenced by these data. However, there are no other data and reports from government officials involved in the process at the time, and some limited after-the-fact verification indicates less gaming than feared.¹⁷

The second problem in estimating abatement exists independently of the quality of the historical data. When the constraint is set a few percentage points below expected BAU emissions, variations in the factors determining emissions can cause the constraint to be nonbinding or for the required abatement to be several times the expectation. For instance, if

¹⁷See note 6.

the cap is set at a level 2 percent below expected BAU emissions and the ex ante distribution of possible outcomes indicates variations of 4 percent from the expectation, the resulting abatement could be zero or three times what was anticipated. A similar problem exists when there is bias in the base from which projections of BAU emissions are made. For instance, in this example, correction for an upward bias in the baseline of 2 percent would effectively wipe out the expected abatement.

Despite these difficulties, several observations can be made. First, and most importantly, the persistently high price for EUAs in a market characterized by sufficient liquidity and sophisticated players must be considered as creating a presumption of abatement. It would be startling if power companies did not incorporate EUA prices into dispatch decisions that would have shifted generation to less emitting plants. There is plenty of anecdotal evidence that this was the case, and the prominent charges of windfall profits assume that the opportunity cost of freely allocated allowances was being passed on (without noting the implications for abatement). Similarly, it would be surprising if there were no changes in production processes that could be made by the operators of industrial plants.

In addition to the presumption of abatement shown above, verified emissions in 2005 are 3.4 percent below the total obtained by adding up the historical “baseline” emissions collected during the allocation process. Moreover, trends in the growth in real output and in carbon intensity point to increasing CO₂ emissions. The growth in real output of the EU economy has been solidly positive, with real GDP in 2005 slightly more than 5 percent above the level in 2002 and similar growth in real output of those sectors included in the EU ETS. The decrease in CO₂ intensity in the EU has slowed noticeably since 2000. From 2000 through 2004, CO₂ intensity in the EU declined at a rate of 0.6 percent, considerably less than the decline of the 2.2 percent rate that characterized the preceding five years. As a result, the rate of increase in CO₂ emissions accelerated from 0.6 percent in the late 1990s to 1.2 percent from 2000 through 2004, despite a lower rate of increase in real GDP. Extrapolating these rates would suggest an increase of counterfactual emissions in the order of 3.5 percent from 2002 to 2005, which would indicate abatement of about 7 percent if the data collected to establish baseline emissions can be considered an accurate reflection of historical emissions. If the bias in the baseline is as much as 7 percent, no abatement would be indicated; but then an explanation for the persistently positive and high price must be developed.

Although better estimates of abatement are possible based on a more country- and sector-specific analysis, any reasonable analysis of the emissions data for 2005 and subsequent years is likely to show that there has been some reduction of emissions attributable to the EU ETS. Although the amount of abatement is likely to be modest, so is the emission reduction goal of the EU cap in this initial, “trial” period.

Conclusion

As noted at the beginning of this article, the EU ETS has succeeded in imposing a price on CO₂ emissions. This is more than can be said for any other country or group of countries. As such, it is by far the most significant accomplishment in climate policy to date. It is too early to provide a definitive assessment of the effects of the EU ETS on producers’ and consumers’ choices, not to mention economic activity, trade patterns, and innovation; but

what has been accomplished in establishing the system, including the nontrivial tasks of deciding caps and allocating allowances, supports some concluding observations.

Global Implications of the EU ETS

The founding governments of the EU ETS hope that it will turn out to be the cornerstone of an eventual global climate regime, but the linking issues raised in the article by Kruger, Pizer and Oates (2007) will have to be dealt with if this hope is to be realized. Still, the EU ETS is creating a reality that will be hard to ignore in future climate negotiations. Just as it is hard to imagine a global climate regime without the participation of the U.S., so also, it is difficult to envision one that does not recognize the EU ETS. The really fascinating issue is how this example will influence the nature of the global regime.

From a global perspective, the most significant aspect of the EU ETS is that it is a multilateral trading system among sovereign nations. Although these states belong to the European Union, the sovereignty retained is still very large. And, although they are all European, they are also very different in economic circumstance and commitment to climate policy, even among the EU15. The differences may not be as pronounced as on a global scale, but the East–West divide in Europe bears some similarity to the global North–South divide. Yet, these differences in circumstance and commitment have been overcome, not so much by adherence to some shared European ideal, as by the combination of desirable community and practical advantage that membership in the EU offers, and of which the EU ETS is one small part. Moreover, the number of countries participating in the EU ETS is increasing. Romania and Bulgaria will join the EU ETS in 2007 as part of their accession to the EU, and it is widely expected that the emissions trading system to be established by Norway, a nonEU member, will be formally linked to the EU ETS.

The potential global influence of the EU ETS goes beyond its role as an example of a multinational trading system. The access to external credits provided by the Linking Directive has had an invigorating effect on the CDM and more generally on CO₂ reduction projects in developing countries, especially in China and India, the two major countries that will eventually have to become part of a global climate regime if there is to be one. The opportunity to sell CO₂ reduction credits to the members of EU ETS does not mean that these developing countries will accept constraints on their CO₂ emissions, but it certainly predisposes them to viewing trading regimes as a desirable global architecture for climate change policy. More will be needed to create the community of interest and practical advantage that would cause countries such as China and India to accept meaningful constraints, but the experience of the EU ETS shows that it can be done.

Is CO₂ Different?

The EU ETS is similar in basic concept and operation to other cap and trade systems, but it is different from all others in being applied to CO₂. The inevitable question is whether this circumstance makes a significant difference. It is also too early to make a definitive judgment here. There is, however, a widely shared perception that the main difference between CO₂ and conventional pollutants is not the obvious chemical composition, but rather the abatement potential. It is pointed out that scrubbers were a demonstrated

technology when the U.S. SO₂ system was implemented and that substitution among coals of widely varying sulfur content was possible. For CO₂, the assumption is that, aside from the often-cited possibility of switching from coal to natural gas in electricity generation, very little abatement is possible except by reducing output. This view is inconsistent with the equally widespread perception that further energy conservation is obtainable at modest cost. But the perception of limited or costly abatement seems to have influenced both the ambition of the caps set in the EU ETS and several of the allocation choices. Time will tell whether the significantly positive carbon price now being paid in Europe will elicit more abatement at low cost, perhaps in unanticipated ways, as has been the case in the U.S. cap and trade programs. The perception of little CO₂ abatement potential also underlies the significant concerns about effects on macroeconomic performance and international trade that have been largely absent from discussions about caps on conventional pollutants.

Toward a More General Principle of Allocation?

The experience with the EU ETS also raises the issue of whether a broader principle is involved in the allocation of allowances. Allocation in the EU ETS provides one more example that, notwithstanding the advice of economists, the free allocation of allowances is not to be easily set aside. Although considerations of compensation for sunk costs and lobbying play a role in this result, there are some notable examples in the European experience that point to a broader principle that may not be any more equitable or efficient, but which seems at least to fit the facts.

As frequently noted (for instance, see Tietenberg 2006), tradable-permit systems are simply one form of limiting access to a common-property resource by issuing usage rights. In the vast majority of these instances, whether they be fisheries, grazing lands, water, or air emissions, the newly created tradable usage rights are granted freely to incumbents who are also, overwhelmingly those who will exercise their new rights after the constraint is imposed. Raymond (2003) perceptively notes that it is as if there is a social norm operating that accords great weight to prior use and the status quo ante when rights that were previously freely exercisable are now limited and made transferable. This has certainly been the case with the U.S. air permit trading programs, as well as in nonair programs throughout the world. The European experience provides further support of this social norm, but with a new twist. The new rights under the EU ETS are contingent on production: recipients ceasing to produce the underlying CO₂-emitting goods forfeit their rights, and new entrants gain rights by the act of production. The qualification introduced into permit trading by the EU ETS is that while the rights are freely transferable, they are not separable from production. It is also too early to say whether this feature will become a characteristic of CO₂ allocations in general or is simply a quirk of the EU ETS, but it is certainly worth noting.

A passing comment in Coase's *Theory of Social Cost* (1960) provides a starting point for understanding this feature of permit trading systems:

If factors of production are thought of as rights, it becomes easier to understand that the right to do something which has a harmful effect. . . is also a factor of production.

Coase was careful not to prescribe how these new rights should be distributed and he emphasized that more than economics was involved. Existing systems, including the EU

ETS, have provided examples of how they are distributed, but it is not clear whether these examples reflect an established, if unrecognized, principle or simply the absence of a better and more politically compelling idea.

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Market and Price Developments in the European Union Emissions Trading Scheme

Frank J. Convery* and Luke Redmond**

Introduction

Every profession has its *idée fixe*, its core response and solution to problems. For example, to address pervasive traffic congestion on roads, the engineer proposes structural solutions, planners favor zoning, and architects see design as the answer. The economist sees prices as the solution. The “right” price will properly reflect the scarcity of road space and bring demand and supply into equilibrium. As a result, traffic will flow and travel time will be saved, shattered nerves will be made whole, road rage will be a distant memory, meetings will begin and end on time, and excuses for being perennially late will no longer be viable.

Much of our work in environmental economics—as researchers, teachers, and practitioners—is focused on trying to produce this “Eureka” moment, when scarcity is appropriately recognized via a market-clearing price. But in the case of climate change and other environmental challenges, the market will not produce the “right” price. Market failure occurs because for exchanges to happen, assets must be owned, they must be divisible, and appropriate legal and institutional mechanisms must be in place to enable a price to emerge. Thus far, in regard to climate change, none of these conditions has applied. The capacity of the earth’s atmosphere to absorb human-caused greenhouse gases is not owned, it can’t readily be divided up and sold, and legal and institutional frameworks have not existed to enable exchange. The market has failed to reflect scarcity value, and global warming pressures have intensified.

In the early 1990s, the European Union (EU) undertook an ambitious effort to provide a price signal through the introduction of an EU-wide tax on energy and carbon. This effort failed because fiscal measures in the EU require unanimous approval by all member states. Such unanimous approval was not forthcoming then, with fifteen member states, and it is even less likely now, with twenty-five member states.

The purpose of this symposium is to increase understanding of a more recent ambitious effort by the EU to correct for the market failure that surrounds climate change. The EU Emissions Trading Scheme (EU ETS) tries to address the reduction of emissions of carbon dioxide (CO₂) by allowing energy-intensive industrial plants and electric utilities in EU

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member states to trade rights or permits to emit CO₂. The units of CO₂ that can be traded are called allowances, or, more specifically, European Union Allowances (EUAs).¹ Through this trading scheme, the EU has succeeded in producing a price signal—via the trading price for EUAs—that does, in some limited sense, reflect the scarce capacity of the earth’s atmosphere to absorb more greenhouse gas emissions.

This paper focuses on the key factors that have influenced the development and functioning of the EUA market that has emerged under the EU ETS. To provide the background and a point of reference for discussion of the EUA market, we first describe the institutional and legal framework for the EU ETS, the scheme’s main design features and provisions, and how these factors are likely to affect the EUA market. Next, in the section on development and operation of the EUA market, we discuss how the EUA market has developed and operated since the establishment of the EU ETS. In particular, we describe the intermediaries that have entered the market to facilitate EUA trading and how the size and frequency of trades have changed over time, identify the key factors that appear to have affected EUA prices, and discuss the trends in EUA trading volumes and prices. The section “Reflections on the Outlook for the EU ETS” offers reflections on the outlook for the EU ETS in the Kyoto Phase and beyond as well as some preliminary conclusions.

The Institutional Framework and Major Design Features of the EU ETS

In this section we examine the features of the EU ETS that shape emissions trading and therefore price. These include the institutional and legal framework of the EU ETS, the nature and scope of the allowance allocation process, and other key provisions of the EU ETS.

The Legal and Institutional Framework

In 1998, following the negotiation of the Kyoto Protocol, the European Commission first broached the idea of a European trading regime in a communication, *Climate Change—Towards an EU Post-Kyoto Strategy* (European Commission 1998). 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* (European Commission 2000), which explained how such a system might operate, and thereby launched the debate within Europe on the suitability of such a scheme. Thereafter, emissions trading was regularly included in the *European Climate Change Programme* (ECCP) as one of the possible measures for meeting the EU’s Kyoto targets.

The Emissions Trading Directive provides the legal foundation for the EU ETS. The European Commission published a draft proposal in October 2001 and it was formally enacted in October 2003; there was to be much discussion and amendments in the two years separating the formal draft proposal and final enactment. The release of the proposed EU Directive on greenhouse gas (GHG) emissions trading in October 2001 initiated the

¹The EUA is the official unit of the European Union Emissions Trading Scheme. Each allowance permits the holder to emit one metric ton of CO₂.

codecision process, whereby the European Parliament and the European Council would review, amend, and approve or reject the directive. The first reading of the proposal took place in the European Parliament in October 2002 and it led to an amended proposal from the commission in November 2002. In March 2003, the Council of the European Union adopted a common position on the directive.² Subsequent agreement between the Parliament and the council led to final adoption of the proposal by the council on July 22, 2003 and promulgation as Directive 2003/87/EC on October 13, 2003.

From early on in the development of the Emissions Trading Directive, the commission had considered the adoption of a specific provision linking Joint Implementation (JI) and Clean Development Mechanism (CDM)³ credits to the EU ETS. As soon as a substantive agreement had been reached on the Emissions Trading Directive, the commission issued the proposal for the so-called “Linking Directive” (European Commission 2003) on July 23, 2003. The Linking Directive entered into force as Directive 2004/101/EC on November 13, 2004.

Technically, the Emissions Trading and Linking Directives would not have legal force for affected sources until they were implemented by national legislation in the member states. In addition, each member state was required to develop its own national allocation plan (NAP), whereby it determined a national total of EUAs and specified the distribution of that total subject to the commission’s review and approval. The NAP process was central to the development of the allowance market because it simultaneously created the potential supply of, and demand for, allowances in the emissions trading market. Given the tight time schedule, very little was in place by 2005. This meant that when the EU ETS became effective, there was considerable uncertainty concerning such fundamental features as the level of the aggregate EU cap and when all member states would participate. These uncertainties were not resolved until well after the EU ETS began.

Caps on Emissions

Under the EU ETS, member states face two “caps” or limits on their CO₂ emissions in the Kyoto period; this does not apply during the pilot phase of EU ETS. The first is “the Kyoto cap,” which comes from the 1997 Kyoto Protocol and applies to the country overall. Under the Kyoto Protocol, the twenty-five EU countries have agreed to a cap on annual emissions to be achieved by 2008–2012.⁴ For the EU15, a burden sharing agreement (BSA) was reached, which distributed among the then fifteen member states the overall reduction of eight percent (from a 1990 base) in annual emissions agreed at Kyoto. Those EU15 members that had less developed economies received increases in their caps, while the more developed economies accepted decreases. For the EU10, their caps comprise the ceilings that they agreed to individually at Kyoto. The second cap is the EU ETS total, which is a “cap

²The text of the Common Position can be found on the commission’s home page: http://ec.europa.eu/environment/climat/emission/history_en.htm.

³JI and CDM are flexible mechanisms under the Kyoto Protocol for recognizing verified project-based reductions in greenhouse gas emissions in developed and developing countries, respectively.

⁴In 1997, the European Union included fifteen member states (EU15)—Austria, Belgium, Denmark, Finland, France, Germany, Greece, Ireland, Italy, Luxembourg, the Netherlands, Portugal, Spain, Sweden, and the United Kingdom. In May 2004, ten more states (EU10) joined the EU—Cyprus, Czech Republic, Estonia, Hungary, Latvia, Lithuania, Malta, Poland, Slovakia, and Slovenia.

within a cap.” This is the quantity of allowances that are allocated to the trading sectors in each country. Thus, for the Kyoto period (2008–2012), the larger the EU ETS cap, the fewer the allowances available under the “Kyoto cap” to cover emissions in the nontrading sectors.

Price Impacts of the Linking Directive

As described above, the Emissions Trading Directive is aimed at implementing the EU’s commitments under the Kyoto Protocol. The Linking Directive recognizes the Kyoto Protocol’s CDM and JI credits, known as certified emission reductions (CERs) and emission reduction units (ERUs), respectively, for compliance purposes under the EU ETS. The Linking Directive underwent several important changes as it evolved from the commission proposal in July 2003 to the final document of November 2004. One of the most important was to allow firms to use CDM credits for compliance in the first, or pilot, trading period (2005–2007) instead of beginning in 2008 as was initially proposed and as is the case for JI credits.

Most firms will aim to meet their emissions obligations at minimum cost. This means that at the margin, adjusting for differences in real and transactions costs, the costs to them of purchasing an EUA under the EU ETS and an equivalent CER will be equal. To the extent that CERs are available at lower cost than EUAs, this will put a downward pressure on the price of EUAs. Linking will increase liquidity and the range of compliance options within the EU scheme, which will likely lead to a reduction in the price of carbon in the EU (i.e., EUAs will become cheaper). However, we would also expect the extra demand for CERs to raise their price, which would likely result in some convergence between the CER and EUA markets.

Another important issue is that, from 2008 onwards, installations in the EU ETS will be competing for CERs and ERUs with governments from around the world, including their own governments, who will be using them to comply with their national Kyoto targets. In addition, they will be competing in the second, or Kyoto, period (2008–2012) with private companies in other industrialized countries that have ratified the Kyoto Protocol, such as Japan and Canada. The use of CDM and JI credits must respect the principle of supplementarity—achieving a significant proportion of one’s emissions reductions domestically. There is no limit on the use of CERs during the pilot period, but there is a requirement that member states limit their use of CDM and JI credits in the Kyoto period to a certain proportion of their allowance allocation, which is to be specified by each member state in its NAP. member states are currently submitting their NAPs for the second trading period, so there is only limited information on the restrictions member states intend to impose on CDM and JI credits. According to the information contained in a number of draft NAPs, the United Kingdom, Italy, and Germany plan to limit the use of CDM and JI credits to ten percent of overall allocations (WWF 2006). Poland and Spain are considering limits of twenty-five percent and fifty percent, respectively.

What are the likely price impacts of the Linking Directive? They are likely to be irrelevant in the pilot phase, because there will be a very limited supply of CERs available in this period, and, adjusting for risk, the price at which they will be available will be comparable to what EUAs cost.⁵ In the Kyoto period, Buen (2006) estimates that the supply of CERs

⁵Buen (2006) reports that the total volumes of CERs available over the pilot period ranges from about 90,000 metric tons in 2005 to about 200,000 metric tons in 2007, compared with a total volume of

will increase to about 200,000 metric tons annually, or 1 billion over the period. We have no estimates of the supply of ERUs (JI credits). It is plausible to conclude that, given the relatively modest magnitude of project-based supply available, the ceilings that have been proposed in the draft NAPs noted above will not constrain purchases. The limitation will be availability of supply, and the price effects will be modest.

NAPs and Registries

Before companies can receive their allowances and begin to trade in the spot market (in contrast to the forward market), the member state's NAP must be approved by the commission and the country must establish an electronic registry. In order for a company to be issued its annual allowance allocation, it must hold an account, known as an "operator holding account," in its national emissions trading registry.⁶ The use of a registry system ensures the accurate accounting of the issuing, holding, transfer, and cancellation of EUAs.⁷ All registries are overseen by a central administrator at the EU level who, through the community independent transaction log (CITL), checks each transaction for any irregularities. The registry system keeps track of the ownership of allowances in the same way that a banking system keeps track of the ownership of money. The directive states that all participants must have been issued their allowance allocations for each calendar year by no later than February 28 of that year. Where there are delays in either the approval of the NAP or the establishment of the registry, the demand and supply of EUAs from the companies in these countries is in effect sterilized. When, as in the pilot phase, these delays occur mostly in countries where supply may be expected to exceed demand, such delays initially put an upward pressure on EUA prices.

Initially, the commission envisaged that by the end of June 2004, it would be in a position to announce its decisions regarding the approval of the EU15's NAPs and that one month later it would announce its decisions regarding the EU10's plans, thereby closing the allocation process. In reality, the commission's NAP approval process was subject to considerable delay because of late submissions by many member states.⁸ This forced the commission to approve member state NAPs in different batches between July 2004 and June 2005. This meant that by February 28, 2005, only three member states (Denmark, the Netherlands, and Finland) had their registries established and had issued allowances to their installations. When the commission released the 2005 carbon dioxide emissions data and compliance status for member states in May 2006, twenty-one member states had fully

allowances in the pilot phase of EU ETS of about 6.5 billion metric tons; he estimates that the price of CERs per metric ton of CO₂ when the seller is taking the risk is in the range €15–19.

⁶It is worth noting that an account may be opened in any of the twenty-five member state registries and that the holding of an account in the national registry is not limited solely to those participating in the EU ETS. Anybody wanting to participate in the allowance market can apply to be issued an account.

⁷The transfer of allowances refers to the transfer both within and between member states. The first provision—transfer within member states—is essential to maintain the integrity of the system. The second is essential to ensure that the market is as wide as possible, so that maximum benefit can be derived from differences in abatement costs.

⁸Austria, Denmark, Finland, Germany, Ireland, and the Netherlands are the only countries that met the deadline.

functioning registries. Finally, the Polish registry, with approximately ten percent of the EUAs, went online in July 2006, a year and a half after the start of the program.

Nature and Scope of the Allowance Allocation Process

The EU ETS is a “bottom-up” scheme, with each of the twenty-five member states making the allocations, establishing and operating the registry, and enforcing compliance. There is wide variation in both the commitment and capacities of the member states. As member states attempt to meet their own self-interest within the constraints imposed by the Emissions Trading Directive, they influence price by shaping the quantity and timing of allowances coming to market and the incentives that firms face to abate, absorb the costs, or pass them through to consumers (which feed back to influence the demand for allowances). The bigger a member state’s share of total allowances, the bigger its influence on price. The five countries that received the largest allocation of allowances for 2005–2007 (in million metric tons of EUAs) are Germany (1497), the UK (736), Poland (717), Italy (697), and Spain (523) (European Commission, www.europa.eu/environment/climat).

During the pilot phase, the EU ETS only covers CO₂ emissions from large emitters in the heat and power generation industry and in selected energy-intensive industrial sectors. This is called the trading sector. It is estimated that the EU ETS applies to installations that will account for 45 percent of CO₂ emissions and 30 percent of total greenhouse gas emissions in 2010 (European Commission 2005).

A size threshold based on production capacity or output was used to determine which installations in the covered sectors participated in the scheme. In the pilot phase, participation is limited to CO₂ emissions from combustion installations with a rated thermal input in excess of 20 megawatts (except municipal or hazardous waste incinerators): oil refineries; production and processing of ferrous metals; manufacture of cement (capacity >500 metric tons per day); manufacture of lime (capacity >50 metric tons per day); ceramics, including brick and glass; and pulp, paper, and board (>20 metric tons per day). Depending on the extent of their allowance allocation, likely emissions, and abatement options, plants will have something to sell or a need to buy. While these initial allowance holders are the key players in price making, as discussed in the section “The Development and Operation of the EUA Market,” third parties are also allowed to buy and sell without restriction. Thus the scope of the market is wide.

The Emissions Trading Directive requires that installations participating in the trading scheme report their actual CO₂ emissions for the calendar year to their respective national authorities by March 31 of the following year. All emissions reports must be approved by an independent verifier. Installations are then given until April 30 to ensure that they have a sufficient quantity of allowances in their national registry accounts to cover their verified CO₂ emissions for the previous calendar year, which indicates their compliance with the EU ETS. The annual compliance cycle of the EU ETS closes with the publication of emissions data and surrendered allowance information on May 15, together with the cancellation of surrendered allowances, which must occur by June 30.⁹

⁹Ellerman and Buchner (2006) provide a detailed discussion of the data on CO₂ emissions and compliance for 2005, which were released by the European Commission on May 15, 2006.

Firms whose emissions exceed the allowances they hold at the end of the accounting period must pay a fine. During the pilot phase, a fine of €40 is applied for each metric ton of CO₂ emitted for which an allowance has not been surrendered. During the second phase of trading (2008–2012), the fine will rise to €100. In either case, those fined must also make up the deficit by buying the relevant volume of EUAs.

Price Impacts of Allocations

A number of factors have created pressure for member states to be generous with their allowances, which puts downward pressure on EUA prices. For example, because the allowances are free, there is a strong incentive for companies in the trading sectors to maximize the quantity of their allowances. In addition, in making their allocations, countries wanted to ensure that they would not be at a disadvantage in competing for new inward investment and that capacity expansion would not be inhibited. This concern also resulted in pressure to be generous with allowances, especially to companies in sectors that appeared to be competing against companies that did not face carbon constraints under the Kyoto Protocol.

It is also important to note, however, that if Kyoto-constrained governments—those that are unlikely to meet their Kyoto targets under business as usual—are very generous in their allocations to the trading sectors, they will have to use some combination of achieving reductions in the nontraded sectors and buying allowances internationally in order to meet their Kyoto cap. Thus, there is an asymmetry in the incentive system, because those companies covered by the EU ETS have a clear and immediate incentive to engage in focused lobbying, while for those outside, including households, the incentives are no less real, but more diffuse.

Another factor that creates pressure for more generous allowances is the fact that the allocations are being made “now,” while compliance with the Kyoto cap is not mandatory until 2008–2012. In deciding whether to approve a member state’s NAP, the commission faces two countervailing pressures—the need to achieve environmental effectiveness through a price that provides incentives for action, which implies “tight” allocations, and the desire to get the scheme operational as soon as possible. The net effect of these incentives would seem to encourage generous allocations, and this in turn would be expected to put downward pressure on EUA prices. It is important to bear in mind that the key objective of the pilot phase was to get the scheme up and running, and to create the institutional and operational structures that would enable the “full” scheme to be fully and effectively operational for the Kyoto (2008–2012) phase and thereafter. Hence, making progress in regard to emissions reduction was not the primary objective in the pilot phase, while it has emerged as a priority for the Kyoto (2008–2012) phase.

The European Commission has indicated its intention to make the 2008–2012 allowance totals lower than the 2005–2007 totals; it made a significant first step in that direction with a decision on November 29, 2006, that requires that the totals proposed by ten member states comprising 42 percent of the EU ETS emissions be reduced to a level that is more than 12 percent lower than the first period totals for those ten member states and more than 7 percent lower than their 2005 verified emissions (see detail in Table 1). There will be considerable economic growth over the 2005–2010 period. Since, other things being

Table 1 Annual allocation of allowances in the pilot and Kyoto phases of the EU ETS, selected countries (in million metric tons of CO₂)

| Country | (1) Annual allocation, pilot phase 2005–2007 | (2) Annual verified emissions, pilot phase 2005 | (3) Proposed annual allocation by member state for 2008–2012 | (4) Decision by European Commission on an allocation for 2008–2012 period | (5) Allowances for additional Installations included in 2008–2012 | (6) Commission decision net of additional installations (4)–(5) | (7) Net commission decision for 2008–2012 as percentage of verified emissions 2005 (6) / (2) × 100 | (8) Economic growth anticipated in 2005–2010 (%) |
|------------|--|--|--|--|--|---|---|--|
| Germany | 499 | 474 | 482 | 453.1 | 11 | 442.1 | 93.3 | 9.6 |
| Greece | 74.4 | 71.3 | 75.5 | 69.1 | 0 | 69.1 | 96.9 | 19.9 |
| Ireland | 22.3 | 22.4 | 22.6 | 21.1 | 0 | 21.1 | 94.2 | 27.2 |
| Latvia | 4.6 | 2.9 | 7.7 | 3.3 | 0 | 3.3 | 113.8 | 50.0 |
| Lithuania | 12.3 | 6.6 | 16.6 | 8.8 | 0.05 | 8.75 | 132.6 | 37.4 |
| Luxembourg | 3.4 | 2.6 | 3.95 | 2.7 | 0 | 2.7 | 103.8 | 27.2 |
| Malta | 2.9 | 1.98 | 2.96 | 2.1 | 0 | 2.1 | 106.1 | 11.9 |
| Slovakia | 30.5 | 25.2 | 41.3 | 30.9 | 1.7 | 29.2 | 115.9 | 32.4 |
| Sweden | 22.9 | 19.3 | 25.2 | 22.8 | 2 | 20.8 | 107.8 | 16.6 |
| UK | 245.3 | 242.4 | 246.2 | 246.2 | 39.5 ^a | 206.7 | 85.3 | 14.3 |
| Total | 917.6 | 868.68 | 924.01 | 860.1 | 54.25 | 805.85 | 92.8 | |

Sources: European Commission, http://ec.europa.eu/environment/climat/ip_1650.htm, DG TREN, "Energy and Transport Trends to 2030—update 2005," and DG ECFIN Economic Forecasts, autumn 2006. Columns (1) through (4) and (8) from tables in European Commission press release at http://ec.europa.eu/environment/climat/ip_1650.htm. Column (5) from footnotes in European Commission press release. Column (6) is the commission decision, adjusted for the additional installations included in the 2008–2012 phase and the inclusion of those in the UK that opted out in the pilot phase.

^aComprised 30 million metric tons from installations that opted out in the pilot phase, and 9.5 million metric tons from additional installations. In the pilot phase, installations covered by the EU ETS were allowed to opt out (i.e., not participate in the EU ETS) provided that they were undertaking equivalent measures to decrease emissions.

equal, such growth will produce rising emissions and associated demand for allowances, the cut is nontrivial. The European Commission is likely to make similar cuts in its decisions on the 2008–2012 allocations for the remaining fifteen member states. This in turn will result in a rise in the price of allowances over the five-year period, which will stimulate further abatement.

Auctioning Provisions

During the pilot phase, member states are allowed to auction up to 5 percent of their total allowance allocation. Thus far, only Denmark, Hungary, Ireland, and Lithuania have exercised this option.

The Irish experience with auctioning illustrates how one member state got involved directly in price discovery. In the Irish NAP, the Irish Environmental Protection Agency (EPA) stated its intention to auction 502,201 allowances. The revenue generated from this auction would be used to defray some of the costs of administering the country's trading scheme. The EPA held its first auction on February 17, 2006. A total of 250,000 allowances, divided into lots of 500 allowances, were made available. With lots set at 500 allowances, this represented a lower threshold for participation in allowance purchasing than most of the allowance market brokers (see "The Development and Operation of the EUA Market"). The auction was open to any person holding a registry account. Participants were allowed to submit up to five mutually exclusive sealed bids. Each bid detailed the price per allowance that the individual was willing to pay, together with the number of lots that they wished to purchase at that price. The EPA received 150 individual bids. Five individual bids were successful in the auction and were offered allowances at a uniform settlement price of €26.30. On the day of the auction, allowances were trading at €26.85 in the brokered market, so there was close to complete allowance price convergence in the two markets. On November 16, 2006, the Irish EPA announced its intention to hold its second allowance auction between December 4 and 5. The major difference between the first and second auctions was that allowances in the second auction were to be sold in lots of one thousand.

Banking and Borrowing Provisions

Allowances are valid for compliance during the period for which they are issued (i.e., the 2005–2007 pilot phase or the subsequent five-year periods). Therefore, although compliance is assessed on an annual basis, banking (the carrying forward of allowance surpluses from one year to the next) and borrowing (using a future year's allowance allocation to offset current emissions) are permitted within each phase. Banking is allowed as unsurrendered allowances are still valid for compliance in the next year(s) of the phase. Borrowing is allowed as allowances for each year are to be issued before February 28, while compliance for the previous year is assessed after April 30. This allows de facto borrowing between January and May, as participants can use this period to make up for a deficit in their holdings.

Although banking is allowed within the pilot phase, with the exception of France and Poland, member states do not allow banking between the pilot and Kyoto phases (i.e., a surplus in 2007 cannot be carried forward to help meet obligations in 2008). This inflexibility is likely to increase price volatility, because as holders of surpluses realize that their surpluses

will have zero value after 2007, they are likely to “unload” at any price. This may result in a price dip. If holders of EUAs could bank their surpluses for use in the Kyoto phase, the price transition would likely be smoother.

Plant Closure Provisions

In the pilot phase, most member states require that if an installation closes it must surrender its allowances. This provision creates an incentive to keep open installations, which might otherwise have been closed, in order to hold onto the allowances. This is both environmentally ineffective and economically inefficient. It also reduces the supply of allowances coming to the market, which puts upward pressure on allowance prices. However, a number of member states, including the Czech Republic, Germany, Greece, France, Luxembourg, Poland, Portugal, the UK, and Spain, permit the transfer of allowances. Under these “transfer rules,” the owners of a closed installation are allowed to transfer their allowances if they are shifting production to another installation that they own.

The Development and Operation of the EUA Market

This section discusses how the EUA market has developed and operated since the EU ETS began on January 1, 2005, and discusses the key factors that appear to have influenced EUA prices.

Intermediaries in the EUA Market

Futures trading in EUAs gathered momentum in 2004. In response, a number of brokers entered the EUA market to facilitate bilateral trades. At the same time, a Norwegian company called Point Carbon began to track and report EUA market developments, including prices.¹⁰ This facilitated price discovery for prospective buyers and sellers. As the market continued to develop, more intermediaries entered the market to meet the rising demand for transactions.

The biggest change concerning intermediaries since January 2005 is that there is now a greater range of market intermediaries. In January 2005, there were seven brokers operating in the market.¹¹ By August 2006, these seven brokers had been joined by five exchanges (a feature not seen in any U.S. allowance markets).¹² The exchanges thus far have not accounted for more than 45 percent of the monthly market activity. Since they first entered the market in March 2005, exchanges have facilitated on average 28 percent of monthly market activity. The total volume of EUA trades will be discussed in more detail in the last part of this section, but Figure 1 shows the smaller-than-expected role that has been played by the exchanges. More specifically, Figure 1 presents the breakdown of weekly traded

¹⁰ Point Carbon, founded in 2000, specializes in analyzing the carbon emissions markets and the power and gas markets. Point Carbon issues reports on its Web site, www.pointcarbon.com.

¹¹ The seven brokers in the market in January 2005 were CO₂e, Evolution Markets, GreenStreamNetwork, GT/SKM Global Environmental Partners, Natsource Europe, TFS, and Vieritas Finance.

¹² The major exchanges that are presently in operation are the European Climate Exchange (ECX), the European Energy Exchange (EEX), Energy Exchange Austria (EXAA), Nordpool, and Powernext.

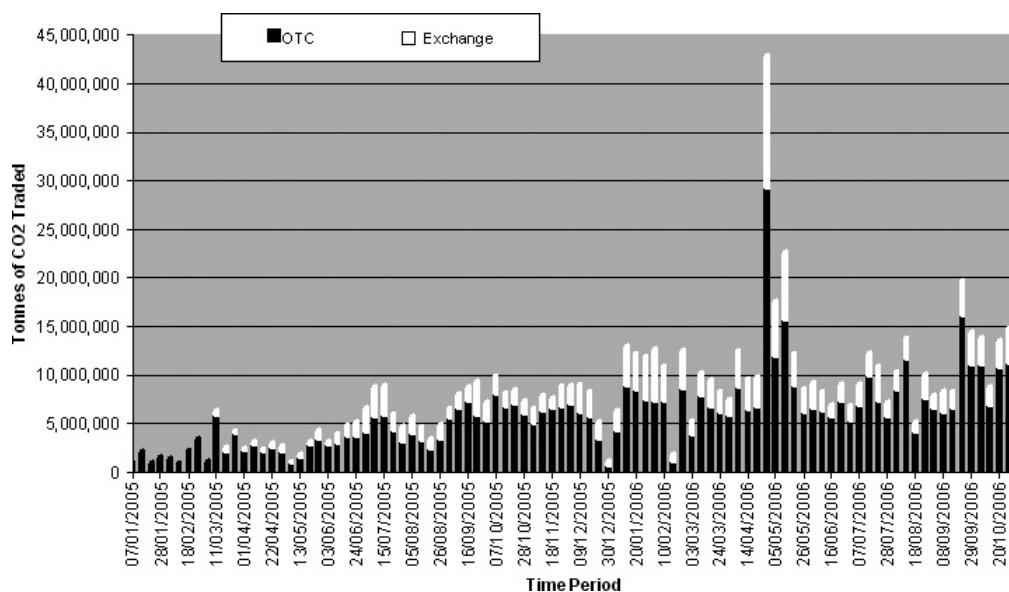


Figure 1. Weekly volume traded in the EUA market (Brokered [OTC], Exchange), January 2005–November 2006. Source: Point Carbon.

volumes between the Over The Counter (OTC), or brokered, market and the exchanges between January 2005 and August 2006.

We conducted surveys of brokers and exchanges in both January 2005 and August 2006 to determine how market operations have evolved since the EU ETS began (see Table 2). We contacted seven brokers and three exchanges. While traders were unwilling to disclose their exact share of market activity, the general consensus was that the European Climate Exchange is facilitating a large proportion of traded activity. The brokers indicated that there were approximately twenty companies that were active in the EUA market in January 2005. We were informed that the majority of these companies were large energy suppliers and large banks, especially German banks. The profile of those most active in the market continues to remain the same, with both large banks (Barclays, Merrill Lynch, BNP, Fortis) and utilities (RWE, E.ON [Munich and UK], Centrica, Scottish Power, Electrabel) being the most heavily involved. While the allowance market continues to be used mostly for compliance purposes and the covering of positions, 2006 has seen the entry of the first potential market speculators, most notably European and American hedge fund managers (Michael Karavias, personal communication, August 21, 2006; Albert De Haan, personal communication, March 29, 2006).

Trends in Size of Trades

When first contacted in January 2005, all but one of the brokers stated that they would not be willing to enter the market to broker a deal for a client unless the client was looking to trade at least 5,000 metric tons of CO₂. This was not a problem in January 2005, since the

Table 2 Conditions and Transactions Costs in EUA Trading in the EU ETS

| | Minimum size of trades (metric tons of CO ₂) | | Transactions cost/brokerage fee per metric ton in euro cents (c) | | Speed of transaction (hours) | | Frequency of trades | |
|--------------------------------------|---|-------------|--|---|---------------------------------|--------------------------------|---------------------|----------------|
| | January 2005 | August 2006 | January 2005 | August 2006 | January 2005 | August 2006 | January 2005 | August 2006 |
| Brokers | | | | | | | | |
| CO ₂ e | 5,000 | 10,000 | ~5c | 1–2% of allowance price | 1–2 hours | Instantaneous | Weekly | Daily |
| Evolution Markets | n.i.a. | 5,000 | n.i.a. | Depends on traded volume and amount of mediation needed between buy and sell counterparts, can range up to 10c per deal | n.i.a. | Instantaneous | Daily | Daily |
| GreenStream-Network, Ltd. | 5,000 | 1,000 | ~5c | Depends on customer profile—1c for interday clients, 1.5–2c for all others | Instantaneous | Instantaneous | Weekly | Daily |
| GT/SKM Global Environmental Partners | 5,000 | 1,000 | 2.5c | Depends on customer profile | Instantaneous | Depends on client requirements | Weekly | Daily |
| Natsource Europe TFS | 5,000 5,000 | 5,000 | ~5–10c 5c | Depends on customer profile ~1–2.5c, dependent on size of deal | Instantaneous Instantaneous | n.i.a. <1 minute | Weekly Daily | Daily Daily |
| Vertis Finance | 1,000 | n.i.a. | >10c | n.i.a. | 2–4 hours | n.i.a. | Weekly | n.i.a. |
| Exchanges | | | | | | | | |
| European Climate Exchange (ECX) | 1,000 | 1,000 | 0.6c (0.2c exchange fee, 0.2c clearing fee, 0.2c delivery fee) | 0.6c (0.2c exchange fee, 0.2c clearing fee, 0.2c delivery fee) | n.i.a. | n.i.a. | Daily | Daily |
| Nordpool | 1,000 | 1,000 | 1c for spot trading 0.5c for forward trading | 1c for spot trading 0.5c for forward trading | Instantaneous | Instantaneous | Daily | Daily |
| Powernext | 1,000 | 1,000 | Depends on customer profile, 2c for regular clients, 3c for infrequent clients | Depends on customer profile, 2c for regular clients, 3c for infrequent clients | 15 minutes | 15 minutes | Daily | Daily |
| European Energy Exchange (BEX) | n.i.a. | n.i.a. | n.i.a. | n.i.a. | n.i.a. | n.i.a. | n.i.a. | n.i.a. |
| Energy Exchange Austria (EXAA) | n.i.a. | n.i.a. | n.i.a. | n.i.a. | n.i.a. | n.i.a. | n.i.a. | n.i.a. |

Source: Author surveys, January 2005, August 2006
n.i.a., no information available.

majority of trades exceeded 10,000 metric tons and these transactions involved large-scale players. However, this raised an important issue. Small- to medium-sized players were expected to enter the market over the coming months. How would they be able to purchase allowances if they were looking to trade less than 5,000 metric tons? Since January 2005, a number of brokers have changed their policies regarding minimum traded volume. Both GreenStreamNetwork and GT/SKM have reduced their minimums from 5,000 to 1,000 metric tons. In addition, the minimum volume for another broker, Vertis Finance, is currently 1,000 metric tons. However, CO₂e increased its minimum from 5,000 to 10,000 metric tons. In general, it appears that the minimum volume requirements of traders discriminates against smaller buyers and sellers.

In order to use the exchanges, a company must be looking to trade at least 1,000 metric tons. However, the traders we interviewed indicated that it is possible to engage in small-volume trades through the EEX exchange.

Third Party Participation

Although, theoretically, any third party can buy and sell allowances, it is not always easy to enter the market. In order for third parties to buy and sell allowances, they must first acquire an account in the registry of any member state. In Ireland, for example, the annual cost of holding an account in the Irish registry is €250. However, once an account has been acquired, it is up to the third party to find a market intermediary to meet its allowance purchasing requirements. Further evidence of the constraints on third parties, especially small players, entering the market is that the authors attempted for several weeks to acquire 100 allowances, without any success.

Changes in Frequency of Trades

The frequency of trades has also changed. When we first contacted brokers in January 2005, only two of them were negotiating trades on a daily basis. As the volume of allowance trade has increased significantly (see figures 1 and 2), it is no surprise that all brokers and exchanges are now active in the market on a daily basis. In addition, the majority of market intermediaries now have a pan-European client base. This is in contrast to the situation in January 2005, when a number of brokers (e.g., GSN and Nordpool) focused on local markets.

Key Determinants of Prices in the EUA Market

CO₂ is quickly establishing itself as a traded commodity, similar to other commodities that are traded on world markets. Here we discuss the key factors that appear to have determined the price of EUAs between December 1, 2004 and July 31, 2006.

The market fundamentals of the EUA market should be the same as for other commodity markets, supply and demand. The main determinants of the supply of allowances were included in the discussions of the NAP process and the nature, scope, and allocation of allowances “The Institutional Framework and Major Design Features of the EU ETS.” The demand for allowances is determined by three factors: (1) the allowance allocations that participants receive (see “The Institutional Framework and Major Design Features of the

EU ETS”), (2) the cost of their carbon-reduction options, and (3) their CO₂ emission levels over the course of the first trading period.

Allowance Allocations

The allowance allocation that installations receive will influence the role that they are likely to play in the market. When an installation’s allocation is generous relative to its emissions, it is highly plausible that such installations will fulfil the role of sellers in the allowance market. Participating installations will be allowance buyers when their allocation is insufficient to cover their emissions needs and/or the costs of abatement exceed the allowance price. It is likely that installations that receive a sufficient allocation will have a neutral role in the market. It is important to note that buyers and sellers in the market will be determined as much by unpredictable variations in output as by whether the past allocation was generous or insufficient. So who will be the main buyers and sellers of allowances in the market? Only an in-depth analysis of individual NAPs will reveal exactly who the sellers and buyers of allowances in the market will be. Central Eastern Europe is expected to be a major seller of allowances. This is because the accession countries, which joined the EU in May 2004, are all well on course to meet their Kyoto targets (as a result of economic contraction in the early 1990s which resulted in reduced emissions) and were therefore able to be relatively generous in allocating allowances to their companies; many in the EU15 will be buyers.

Market Power

Individual companies may have market power—the ability to influence price by managing the supply of and/or the demand for a product or service. A sense of potential market power in the allowance market can be judged by the share of total allowances “held” by companies in each of the trading sectors. The ten companies that were allocated the largest number of allowances under the EU ETS are presented in Table 3. The total allocation that a company received is the sum of allocations allocated to each of the individual installations owned by a company.

Another indicator of market power is the Herfindahl-Hirschman Index (HHI).¹³ In this case, the market refers to the EU allowance market that was created as a result of the twenty-five member states allocating approximately 6.5 billion allowances amongst almost 11,500 installations. Calculation of the HHI for each of the top ten companies in Table 3 indicates that at the individual company level there is no evidence of any company having market power in the allowance market. This conclusion is not very surprising given that the company with the largest share of allowances still only accounts for 5.9 percent of total allowances. Since seven of the top ten companies are in the electricity-generating sector, we calculated the HHI for the electricity-generating sector to determine whether there is any market power

¹³The HHI is a measure of the relative size of firms in relation to a market and indicates the amount of competition between firms. The HHI computes market power based on the percentage share of a total market. The HHI can range from 0 to 1. The higher the HHI, the more concentrated is the market power, with a small index indicating a competitive industry with no dominant players. In general, an HHI below 0.1 indicates an unconcentrated market.

Table 3 Market Power in the EU ETS

| Company | Sector | Allowances allocated (million) ^a | Share of total EU ETS allocations (%) | Herfindahl- Hirschman Index |
|---------------|---|---|---|-----------------------------------|
| 1. RWE | Electricity generation | 386 | ~5.9 | 0.003481 |
| 2. Vattenfall | Electricity generation | 234 | ~3.6 | 0.001296 |
| 3. Enel | Electricity generation | 146 | ~2.2 | 0.000484 |
| 4. E.ON | Electricity generation | 116 | ~1.7 | 0.000289 |
| 5. EDF | Electricity generation | 107 | ~1.6 | 0.000256 |
| 6. Corus | Production and processing of ferrous metals | 92 | ~1.4 | 0.000196 |
| 7. Endesa | Electricity generation | 86 | ~1.3 | 0.000169 |
| 8. E.ON | Electricity generation | 86 | ~1.3 | 0.000169 |
| 9. Shell | Oil refining | 62 | ~0.9 | 0.000081 |
| 10. Arcelor | Production and processing of ferrous metals | 59 | ~0.9 | 0.000081 |
| Total | | 1374 | ~20.8 | 0.006502 |

Source: For allowance allocations, Henrik Hasselknippe, personal communication, August 15, 2006; for HHI, author calculations are based on Point Carbon data.

^aAllowances allocated are in Mt CO₂ for three years, and they are approximations rather than actual allocations. Ownership structures make it difficult to give an exact number.

at the sectoral level. With a HHI of 0.031, we can conclude that the electricity-generating sector does not appear to possess any real market power in the EUA market.

Determinants of CO₂ Emission Levels and EUA Prices

CO₂ emission levels, and hence the demand for EUAs, will depend on a number of factors, including weather, fuel prices (and their effect on fuel switching), and economic growth (Point Carbon 2004).¹⁴ Weather—temperatures and precipitation—can be expected to affect the price of carbon through its impact on the demand for, and supply of, electricity.¹⁵ However, an analysis of the EUA market between December 1, 2004, and July 31, 2006, (Redmond and Convery 2006) reveals that movements in energy prices have had the most significant impact on the development of EUA prices. Of the three energy commodities analyzed (coal, oil, and natural gas), increases in the price of oil appeared to have the biggest estimated impact on allowance price. The authors believe that the price of oil shows up so strongly because it reflects the tying of gas contracts to oil prices. This analysis suggests that weather to date has not been a major contributing factor to EUA price development. The insignificance of weather is surprising, as one would expect the demand for electricity for heating and cooling purposes in response to weather conditions to have an impact on

¹⁴Given that our analysis focuses on a recent and relatively short period, we did not include economic growth in our analysis.

¹⁵Numerous studies, including Engle et al. (1986), Filippini (1995), Li and Sailor (1995), Henley and Peirson (1997, 1998), Considine (2000), Johnsen (2001), and Pardo, Meneu, and Valor (2002) have found that temperature is one of the main meteorological factors that affects electricity demand.

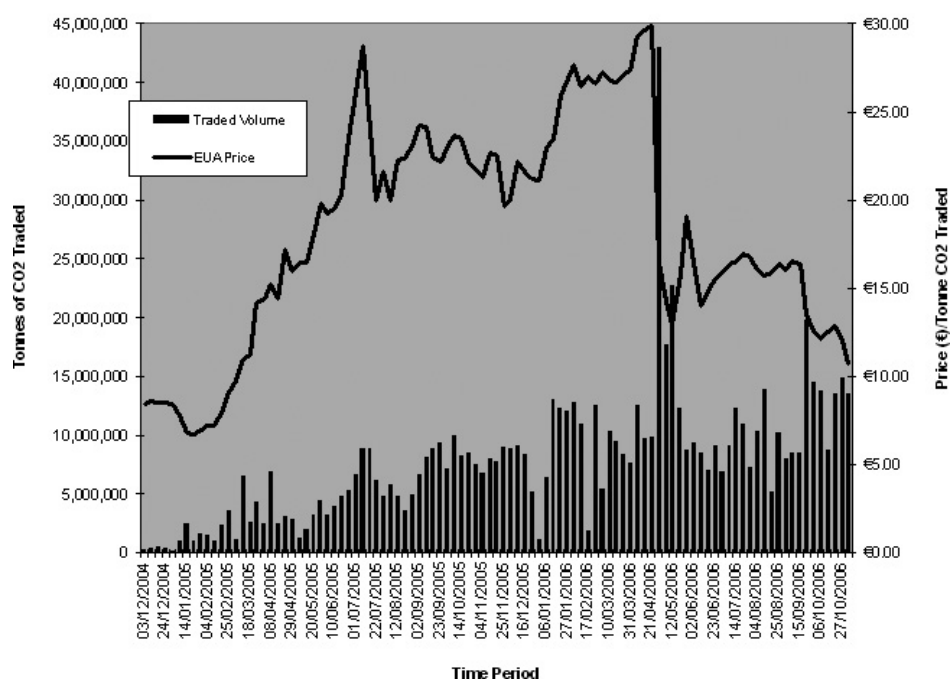


Figure 2. Weekly EUA price and traded volume development. Source: Point Carbon.

the demand for allowances. However, weather may appear to be insignificant because the weather effects tend to be offsetting (Houbert and De Dominicis 2006).

Trends in Volume of EUA Trades and Prices

The allowance market has developed significantly—both in terms of price evolution and traded volume—since the EU ETS officially began in January 2005.¹⁶ As noted above, in anticipation of the establishment of the EU ETS, beginning in 2004, a futures market in EUAs developed, in which it has become the convention that the contract is settled in December of each year. This allowed prices to be quoted and a thin market to exist in early 2004, well before there were any allowances to trade. Initially, trading was on a very small scale, but increased over time. In particular, the volumes transacted accelerated following the official launch of the EU ETS on January 1, 2005. The trends in EUA price and trading volume between December 3, 2004, and November 3, 2006, are shown in figure 2.

On January 3, the price of the 2005 vintage EUA was €8.38. Twelve months later (December 30), the closing price was €21.10. Prices seen in the market over the course of the first official year of the EU ETS surpassed all expectations of market analysts and academics alike. Given a low market price of €6.68 in the middle of January 2005, very few would have imagined that six months later the price of an allowance would have scaled the dizzy heights of €29 and come close to breaking the €30 barrier. But between July 2005 and April 2006, the allowance price consistently traded over the €21–€30 range.

¹⁶While we have adequate information on volume of trades and prices, our information on who is buying and selling is incomplete.

Recent Price Volatility

Since mid-April 2006, the EUA price has been subject to considerable volatility. The market reached a price of almost €30 (€29.90) in mid-April, fell to €12 in mid May, recovered to almost €17 in mid August only to fall back to €8.45 on November 13 (www.pointcarbon.com). The sharp fall in price witnessed in the spring of 2006 has been attributed to the release of actual CO₂ emissions data for 2005. As discussed in “The Institutional Framework and Major Design Features of the EU ETS,” the annual compliance cycle of the EU ETS is closed by the publication of emissions data and surrendered allowance information for installations. The announcement of these market-sensitive data is seen as revealing a true picture of the market, as well as providing a clear indication of the market for the remainder of the trading period. While the commission planned to release this information on May 15, member states were not prohibited from disclosing their own verified emissions. The reality of the situation was that by the time the commission officially announced that the market was long by some 44 million allowances, a price collapse had already occurred, because in the weeks preceding the commission’s announcement, a number of member states released their verified emissions data. Between April 24 and May 2, five member states (the Netherlands, Czech Republic, France, Sweden, and Belgium [Walloon Region]) released information showing that their overall 2005 position was long. The commission deemed that the independent release of this information was having a significant impact on the market and on May 2 asked all member states to refrain from further release of verified emissions data. Nonetheless, rumors relating to long positions in a number of other countries, most notably Germany, continued to circulate in the market. Given the dramatic fall in price in response to the verified emissions data, it is plausible to conclude that not many had expected a surplus in the EUA market. Two possible explanations exist for the overall surplus of EUA. First, installations may have reduced their emissions as a result of improvements and/or investments in energy efficiency or switching to less CO₂-intensive fuel types. Second, the surplus could be a reflection of overly generous allocations by some member states.

Recent Traded Volumes

The data show that approximately 291 million allowances were traded in the EU ETS during its first year of operation. By October 2006, this figure had increased significantly, to approximately 787 million allowances. Assuming an average allowance price of €18.69 over the course of the first twenty-two months of the EU ETS, traded volume to date is valued at €14.7 billion (\$18.86 billion).¹⁷ Prior to the price collapse in spring 2006 there was no sign of any extraordinary market activity. However, once price began its bearish run, market activity picked up, with a large spike in total traded volume—42.9 million allowances—in the week ending April 28, 2006. This spike occurred because companies traded EUAs in order to meet the April 30 deadline for installations to surrender a quantity of allowances

¹⁷The average allowance price was determined on the basis of the weekly allowance price appearing in Point Carbon’s weekly newsletter—Carbon Market Europe (www.pointcarbon.com)—over the period January 7, 2005–November 3, 2006.

equal to their total CO₂ emissions during the previous calendar year. This extraordinary level of market activity helped to ensure that April 2006 became and still is the most liquid month in the history of the EU ETS.

Role of the Power Sector

In the weeks following the commission's official release of emissions data, a paradoxical situation was seen in the allowance market. In spite of the market being long in 2005, with the chance that this position could be replicated for the remainder of the trading period, allowance prices instead of falling began to rise. Analysts at Point Carbon believed the rise in price was caused by the market activity of the power sector.¹⁸ While the allowance market in general was long in 2005, these analysts pointed out that the power and heat sector was short by some 35 million allowances. In spite of the allowance surplus, not much of it was seen in the market place. As a result, the power sector, which has an immediate need to cover its production, was forced to buy allowances at whatever the prevailing market price happened to be. In addition to some power generators that needed to buy, traders pointed to some generators that were buying allowances to "hedge plant production" and "lock in profit."

Since August 2006, the EUA price has been on a steady decline, while the volume of traded allowances has been significant. Total traded volumes for August, September, and October were 37.4 million, 52.9 million, and 57.6 million, respectively. The fall in price over this period has been attributed to a combination of changes in relative oil, coal, and gas prices, and a reduction in the demand for allowances from power generators, who, over the course of this period, have hedged their 2007 power positions and no longer have a strong need to keep supporting the price of EUAs.¹⁹

Conclusions on Market Development

The recent European experience with CO₂ emissions trading yields many interesting insights. Once a decision has been made to create a trading scheme, a futures market develops spontaneously. It is possible to create a market even where there is a degree of confusion and chaos in the allocation and implementation process and where key players—notably Poland in the EU ETS case—are not involved in the market at all initially. It demonstrates how difficult it is to predict allowance prices. The prices throughout most of 2005 were higher than most analysts expected, for a variety of reasons, some of which were discussed above. In addition, one-time "exogenous" events affected prices, such as a long strike in the Finnish pulp and paper sector, which reduced demand for allowances.

It appears that whenever "real" information on the long-short balance of allowances is provided, which happened in the EU in April 2006 and indicated that there was more "supply" relative to demand than had been heretofore envisaged, the market price immediately adjusts, in this case downwards. This pressure was exacerbated by the fact that

¹⁸See Point Carbon, "Is Industry Maintaining High European Power Prices?" *Carbon Market Europe*, www.pointcarbon.com, May 26, 2006.

¹⁹See the October 6, 2006, and November 11, 2006, editions of Point Carbon's *Carbon Market Europe* at www.pointcarbon.com.

April 30 was the final date for installations to cover their emissions for 2005. The spike in allowance price before the sharp fall on April 28, 2006, meant that those who had just previously bought allowances paid a substantial premium for their decision. As economists, we are sanguine about such movements—“it’s the market, stupid”—and winners and losers balance out. However, for those in the market who, for whatever reason, decided that they had to buy and ended up doing so at the top of the spike, it was surely an unhappy experience. These players may provide support for a price cap that if enacted would engender a number of economic and environmental inefficiencies. Given the market sensitivity of emissions data and the sharp fall in prices associated with its release this year, there is a need for the commission to develop a strategy that ensures a more harmonized release of this information that minimizes the impact on the market. There may be a case for interim—perhaps quarterly or semiannual—reconciliation that provides more frequent hard information. Of course, the market will adjust by hedging and buying forward, as is routine now for a variety of energy and other commodities.

Reflections on the Outlook for the EU ETS

What is the EU ETS likely to look like in the future, and will it continue beyond the Kyoto phase?

Future Scope of the EU ETS

Recent developments indicate that the scope of the EU ETS is likely to be extended during the Kyoto phase (2008–2012) and thereafter to include aviation, and perhaps also additional gases. The European Commission (2005) has published a communication that advocates including aviation, and the commission plans to present a legislative proposal by the end of 2006. On November 11 a leaked draft revealed that from 2011 the European Commission is proposing that the aviation sector be included in the EU ETS.²⁰ Under the proposal, the operators of both domestic and international flights landing and taking off on an EU airfield would be required to surrender allowances to cover their emissions. Allowances would be allocated at the EU level to avoid national governments seeking to over allocate their national flag carriers. The draft states that airlines would be allocated “aviation allowances” which could only be used for compliance in the aviation sector (i.e., the aviation sector would be unable to sell to installations in other trading sectors in the EU ETS). Because of a lack of binding commitments for international aviation emissions under the United Nations Framework Convention on Climate Change and the Kyoto Protocol, the parliament wanted to restrict the aviation sector from selling into the ETS. However, airlines would be able to purchase allowances from other sectors to meet their own emissions requirements.

It is notable that while electricity and road transport are the two main sources of growth in greenhouse gas emissions, adding road transport to the EU ETS has not been given

²⁰The leak to Reuters was reported in both the November 15, 2006, edition of the *International Herald Tribune* (pp. 1, 15) and the “Daily News” section of the Point Carbon Web site (www.pointcarbon.com).

serious consideration among policy makers.²¹ This may be because most EU countries already impose substantial excise duties on gasoline and diesel at rates that far exceed any conceivable allowance price level. For example, the per CO₂ metric ton equivalent excise duty on gasoline in Germany on July 31, 2006, amounted to €275.20 (Directorate General Energy and Transport 2006) or more than ten times the allowance price.²² Governments are likely to be very reluctant to give up this revenue in exchange for a (free) allocation of allowances, and the environmental community would likely be fearful of the environmental implications. In contrast, aviation fuel is not subject to excise duties or other substantive taxes, so no revenues would be foregone by including this sector in the EU ETS.

To evaluate the current scheme and plan for the future, the commission has initiated an analysis of the design and functioning of the EU ETS scheme, the impact of expanding it to other sectors and gases, and its impact on competitiveness. In addition, on June 16, 2006, the European heads of state agreed to a renewed EU Sustainable Development Strategy (Council of the European Union 2006). This is an important statement of policy because it sets targets for overall reduction in greenhouse gas emissions by 2020 in the range of 15–30 percent from the 1990 levels, and EU-specific targets for renewables. It also specifically instructs the commission to review the EU ETS and to consider extending it to other gases and sectors. Implicit in all of this is the view that the EU ETS will continue after the Kyoto phase.

There is also likely to be a growing focus on linking the EU ETS with emerging trading schemes in Japan, Canada, and, in time (i.e., the post-Kyoto period), the United States. There will also be increasing interest in linking trading with carbon sequestration and other apparent lower-cost options for reducing the pressure of greenhouse gas emissions.

Outlook for Prices and Trading

Looking to the short-term future, we expect downward pressure on prices towards the end of the pilot phase, as the supply of Polish allowances comes on stream and as companies holding surpluses realize that, with the exceptions of France and Poland, pilot-phase allowances will have no value in the second phase. During the second phase, we expect prices to tighten again as member states make their allowance allocations with the requirements and costs of meeting their overall Kyoto caps in mind and as the commission makes its decisions. The decisions already made concerning the allocations for the ten member states implies a significant reduction in supply relative to the pilot phase. The price effects of expanding the EU ETS to include additional sectors will depend on whether these sectors would be net buyers, upward price pressure, or sellers, downward pressure.

Outlook beyond the Kyoto Phase

After 2012, we expect emissions trading to continue independent of the decisions that are made in regard to global post-Kyoto arrangements. There are several reasons for this.

²¹It has, however, been given some attention by scholars. The debate is summarized in Ryan and Turton (2007).

²²This excise duty is equivalent to €2.48 (\$3.20) per U.S. gallon.

There is significant political commitment within the EU. For example, Miliband (2006), secretary of state for Environment, Food, and Rural Affairs for the UK, recently made a strong case for not just maintaining the EU ETS, but extending it by allocating individual carbon points that can be traded. In addition, no major political party in Europe supports ending the EU ETS. Moreover, administrative bureaucracies are now in place in every member state, which creates a strong vested interest in keeping the emissions trading system in place. Financial intermediaries are making money through the EU ETS, and are interested in continuing to do so.

There is also likely to be considerable support from industry. First, the value of the allowances transferred (applying an estimate of €10–20 per metric ton) is €65–130 billion, an amount that is clearly not trivial. In addition, major global companies such as General Electric, BP, and Shell see the advantages of the EU ETS and have supported it. Industry also realizes that the counterfactual to the EU ETS—what would be put in place instead—could be worse. Most of the complaints about the EU ETS have come from industries that were not included (e.g., the smelter industry). These companies did not benefit from free allowances, but have still had to bear the costs of the ensuing higher electricity prices.

A stakeholder survey was launched in May 2006 as part of the commission's review of the EU ETS.²³ The feedback from stakeholders has been largely positive, with stakeholders seeing the EU ETS as the principal mechanism for addressing climate change. This feedback has also focused on addressing the uncertainties inherent in the short timeframe of the current scheme: the need for targets, the desirability of extending the EU ETS to include new gases, the need for more harmonization across countries in the allocation of allowances, and the need for an international framework within which to operate.²⁴

There are also strategic reasons to continue operating the EU ETS. Because of its vulnerability to energy supply interruptions from the Middle East (oil) and Russia (gas), the EU needs to reduce its dependence on imported fossil fuels. The EU ETS helps achieve this goal by operating at the margin as a pan-EU tax on imports. This makes local alternatives, notably energy conservation and renewables, more commercially viable, which has also made the EU ETS attractive to the renewables lobby.

Finally, as global emissions trading markets emerge, both the operational experience of the EU ETS and the fact that the EUA price will be the lead carbon price quoted in international markets—the equivalent of “Brent Crude”—will create pressure to continue the EU ETS into the future.

Conclusions

The EU ETS is the largest emissions trading scheme in the world. It has been politically feasible for several reasons. First, the unique institutional structure of the European Union gives exclusive rights to the commission to initiate proposals. This means that there was

²³Stakeholders included participating companies, governmental bodies, industry associations, market intermediaries, and NGOs.

²⁴These concerns are summarized in www.defra.gov.uk/environment/climatechange/trading/eu/future/review/questionnaire/htm. This Web site is also a good source for information in English on the latest developments in the EU ETS.

always a single dedicated facilitator and champion of the trading agenda. Second, a culture of acceptability was created in key parts of the policy community through the dissemination of information, which helped the idea of the EU ETS to become part of the policy agenda. This was achieved through informational workshops that were held throughout the EU, which included participants from academia, member state government organizations, nongovernmental organizations, and industry. In addition, the successful U.S. experience with the acid rain program was meticulously documented by Ellerman et al. (2000). Further, researchers at Resources for the Future, the Massachusetts Institute of Technology, and other institutions were willing to act as independent and informed agents for the idea of the EU ETS, and European academics working with the research networks Concerted Action on Market Based Instruments and Concerted Action on Tradable Emissions Permits fostered an informed and supportive culture. Third, unlike the earlier carbon tax proposal, unanimous support from the member states was not required. So it was impossible for any one member state to veto the proposal. Fourth, there was strong support from the European Parliament, which had powers of codecision, and private companies such as BP and Shell, which had initiated their own schemes. Fifth, the UK government's decision to proceed unilaterally provided both a demonstration effect and an urgency to capture the economies of scale and scope that an EU-wide scheme could provide. Finally, the Clinton administration's insistence that emissions trading be included as a flexible instrument in the Kyoto Protocol provided the global platform on which the EU ETS sits.

There were significant one-time administrative costs involved in setting up the EU ETS. Each of the twenty-five member states has had to invest considerable front-end resources to negotiate and make the allocations, set up and manage the registries, determine emissions, establish monitoring schemes, and so on. However, lessons were learned from the U.S. experience with emissions trading, which has helped the EU ETS avoid other administrative costs. For example, no permission is required to make trades; the system is cap and trade, thereby avoiding the costs of negotiating a baseline for each sector. Intermediaries were encouraged to "make" the market, with minimal guidance or interference, such that the costs of buying and selling are very low and there is considerable competition among brokers and exchanges, and banking—and some degree of borrowing—has smoothed the transactions.

As of November 2006, EU member states have allocated approximately 6.5 billion allowances among the installations participating in the EU ETS, and, for the first time, a transnational price exists that signals scarcity and opportunity cost in regard to climate change. Emitters of CO₂ in over 11,000 installations in power generation and energy-intensive industry in the EU now face price signals that reflect the fact that the earth's capacity to absorb more greenhouse gas is limited.

This price signal goes beyond the EU ETS. It also provides a basis for evaluating initiatives in the CDM and JI markets, other trading under the Kyoto Protocol and beyond, and emerging opportunities for carbon sequestration. Given the constraints implied by free allocation, the variable transaction costs are moderate, and the market clears. There is demonstrated political support and institutional capacity to implement the system. Fairness issues, focusing mainly on the CO₂ allowance market's effect on energy prices for more vulnerable members of society, are likely to become more prominent. How fairness issues are addressed may shape the political viability of the scheme going forward.

At the time of writing (November 2006), the EU ETS is only halfway through the pilot phase, so it is premature to make any definitive judgements about the scheme. While it would be easy to construct a hypothetical EU ETS that would have been more efficient and equitable, such a system was not politically feasible. The pilot phase provides a practical platform that can and will evolve into a scheme that is likely to include aviation, other sectors, and other gases. Just as the acid rain scheme in the US provided the template for all subsequent schemes, the EU ETS experience is likely to help shape the global development of emissions trading in greenhouse gases.

Although we are optimistic about the long-term future of the EU ETS, it remains the case that the EU accounts for not much more than twenty percent of the global greenhouse gas emissions, and this share is set to shrink over time. To some extent, the EU ETS represents an act of faith that its leadership will result in a wider constituency for effective action in the longer term. However, unless a global framework emerges out of the current discussions that is “incentive compatible” with key players, such optimism may prove to be misplaced.

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Decentralization in the EU Emissions Trading Scheme and Lessons for Global Policy

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Introduction

The European Union (EU) has created not only the largest emissions trading scheme in the world, but also a system that has an innovative structure with some intriguing and problematic properties. In particular, the EU Emissions Trading Scheme (EU ETS) has its own decentralized character. In thinking about the decentralization issue, we can envision a range of design possibilities. At one extreme is a wholly centralized system in which the central environmental authority determines who will participate in the market, how many permits will be created, how these allowances will be distributed among the various emission sources (that is, installations being regulated), as well as all the rules for compliance and trading. At the other end of the spectrum is a completely decentralized system in which each country (or jurisdiction) runs its own system with no automatic links or connections to other jurisdictions. The EU ETS is midway along this spectrum, with the European Commission (EC) making certain basic decisions concerning the structure of, and participation in, the system, but member states deciding their national cap level, allocating the country's permits—also called allowances—to sources, creating institutions to monitor, report, and verify their emissions, and making choices about some structural features (such as auctions and banking). The EU ETS draws on the U.S. sulfur dioxide (SO₂) trading system for much of its inspiration, but relies much more heavily on decentralized decision-making for the allocation of emission allowances and for the monitoring and management of sources. Because the EU ETS links together emissions trading programs in so many separate countries, it also raises the broader issue of whether linking trading systems in different regions of the world in order to create a more global regime for trading carbon dioxide (CO₂) is ultimately feasible and desirable.

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Our purpose in this article is to examine the structure of the EU ETS, draw out its implications for the operation and particularly the efficiency of the EU ETS itself, and explore how the experience with the EU ETS might inform the design and functioning of any future global emissions trading system. We begin our analysis with a brief review of the structure of a centralized system of emissions trading, a description of the basic structure of the EU ETS, and then a discussion of the economic implications of decentralization and other key provisions of the EU ETS. Next, we discuss the basic analytics of linking emissions trading systems as well as the bigger question of whether linking is always desirable. We then turn to how these linking issues have played out in the context of the EU ETS. Finally, we look at the implications of the EU ETS for the establishment of a broader international emissions trading system.

The Decentralized Structure of the EU ETS

This section focuses on the decentralized structure of the EU ETS and its implications for economic efficiency. To provide a frame of reference or “benchmark” for our discussion, we begin with a description of the standard, more centralized emissions trading system and then point out the specific ways in which the EU ETS is different.

Standard Emissions Trading System: A Benchmark

In the conventional system of emissions trading, an environmental authority imposes a limit or “cap” on total emissions and then issues permits for units of emissions, where the total number of permits equals the cap. In establishing the permit market, the authority makes three fundamental decisions. First, it defines who will participate in the market: the sources of emissions that can buy and sell permits. This effectively determines demand in the permit market, since the demand curve is simply the horizontal summation of the demand curves of the individual sources. Second, when the authority specifies the cap, it sets the supply in the market because this determines the number of permits that will be available.¹ Third, it sets the market in motion by actually allocating the permits among the sources. This can be done in either of two fundamentally different ways: an auction of the permits or a free distribution of the permits among sources or other entities, often by a formula based on historic emissions or output. The authority may also employ some combination of these two methods.

The resulting market equilibrium will determine the actual pattern of emissions among the sources. A cost-minimizing source will purchase (or sell) permits until its marginal abatement cost (MAC) equals the market price of a permit. This implies that, in equilibrium, the MAC of all sources will be equal, which satisfies the necessary condition for minimizing the aggregate abatement cost of realizing the cap. In addition, in the absence of transactions costs, this outcome does not depend upon how the permits are initially allocated among sources, because the actual emissions of the various sources are independent of the initial distribution. Thus, in principle, the emissions trading system generates the least-cost pattern

¹This does not include offsets—credits generated outside the trading system that can be used to meet allowance requirements. We return to this issue later.

of emissions among sources. The system also provides a continuing incentive for research and development (R&D) in abatement technology, since, by finding more effective and less costly ways to reduce emissions, sources can reduce the number of permits they need, thereby increasing their profits.

It is important to note one additional property of the “standard” system: the specific allocation of permits typically has no effect on the output prices of polluting firms. Under free distribution, initial allocations constitute a one-time transfer of wealth to the sources. Each source receives an asset whose value is equal to the number of permits in its allocation multiplied by the price of a permit. As noted above, a cost-minimizing source then buys or sells permits until its MAC equals the price of permits. So, in this general setting, the initial allocation of permits has no effect on either marginal abatement costs or the marginal cost of output.²

In addition to these fundamental decisions, there are a myriad of other choices facing the environmental authority as it creates an emissions market. Two choices are particularly relevant in discussions of decentralization and linking: those concerning flexibility mechanisms over time and compliance. While emissions trading minimizes costs among sources during a particular period of time, costs can still be low in one period and high in another. A banking mechanism allows sources collectively to reduce emissions below the overall cap when costs are low, accumulate excess allowances, and then use them to increase emissions above the overall cap during a future period when costs are high. Although shifting emissions over time has consequences for some pollutants, the exact timing of CO₂ emissions is of little importance in terms of their environmental impact. What matters is the accumulated volume of emissions over a longer period of time (years or even decades). It thus makes economic sense to provide flexibility in the timing of emissions.

While banking allows sources initially facing low costs to over-comply and save allowances for future periods of high costs, it does not address the possibility of high costs in the initial period followed by lower costs. One mechanism for dealing with this possibility is a “safety valve.” A safety valve refers to a specified price for emissions that is well above expectations (known as a “trigger price”) that comes into effect when the market price for permits threatens to become excessively high, signaling a time of inordinately high abatement costs. Sources have the option of simply paying the trigger price for any emissions in excess of their allowances, instead of bidding and raising the price of permits to yet higher levels. The trigger price thus effectively sets a ceiling on marginal abatement costs, thereby allowing sources to maintain their emissions during times when controlling emission is extraordinarily costly. The economics literature has long made a compelling case for such a device, and, as Newell and Pizer (2003) have shown, a sensible safety valve offers large potential savings in the climate change context (such a mechanism has been discussed in the context of actual policies in the United States and Canada; see Government of Canada, 2002; U.S. Senate

²This assumes a setting of competitive markets with marginal-cost pricing. In fact, markets in energy-generating sectors are often regulated markets, where the form of regulation effectively results in some kind of average-cost pricing. In such a regulated setting, the way in which permits are allocated matters for output and pricing decisions. Under an auction of permits, for example, the cost of permits tends to be passed through in the form of higher energy prices.

2005). Borrowing is another alternative mechanism, as well as options combining various borrowing and safety valve features (Doniger, Herzog, and Lashof 2006).

A final set of design choices involves compliance. An environmental authority must decide how sources will monitor, report, and verify their emissions in order to ensure that emissions do not exceed the cap. Such features are often taken for granted in the United States, with its strong legal traditions and watchdog organizations. These features create the foundation for trust, confidence, and fairness in a trading system, but they are less certain in countries with weaker institutions or with different traditions.

The U.S. system for trading SO₂ allowances is one example that ties together all the features of the standard emissions-trading model. This “cap-and-trade” system establishes a cap on overall national emissions for trading sources and a mechanism for allocating the permits that relies primarily on historic levels of heat input and an emissions performance standard. The program allows unrestricted banking of allowances and uses continuous emissions monitors (particularly coal-fired units) to ensure compliance. This has been a carefully studied “grand experiment” in environmental policy-making. As noted by Burtraw and Palmer (2004), it has been highly successful in achieving its goal of reducing emissions at relatively low cost; it has become a model for other cap-and-trade programs.³

The Basic Structure of the EU ETS⁴

The EU ETS differs in some important ways from the standard model. Most significantly, as noted above, it adopts a more decentralized structure of decision-making that leaves much of the authority for key decisions to the member states. This, as we shall see, has some important implications for the way the system operates and its expected efficiency.

The starting point for the EU system is an overall cap on total emissions from all sectors of the economy in all 25 member states that is equal to the EU commitment under the Kyoto Protocol. Given this overall cap, the central EU authority has specified the sectors of the economy—the “trading sectors”—that will initially participate in the EU ETS. This encompasses four broad sectors: iron and steel, certain mineral industries (including the cement industry), energy production (including electric power facilities and refining), and pulp and paper. It is estimated that this includes over 12,000 installations that account for about 46 percent of CO₂ emissions in the EU. Because half of EU emissions remain outside the trading program, the EU’s Kyoto cap necessarily will be met by a combination of efforts by sources in the trading sectors and by controls on sources in the nontrading sectors.

Within the EU-wide Kyoto target, each member state has its own national emissions target as determined under the EU burden-sharing agreement, which defines each member state’s emissions reduction obligation. Each country is required to develop a National Allocation Plan (NAP), which, among other matters, addresses the national emissions target in two steps. First, it allocates the country’s total burden-sharing target between the trading

³For an excellent review of the experience with trading programs, see Stavins (2003). For a comprehensive treatment of the principles and practice of emissions trading, see the new edition of Tietenberg’s classic work (2006). Stavins (2003) and Ellerman et al. (2000) discuss how U.S. trading programs have benefited from the temporal flexibility provided by banking provisions.

⁴For a more detailed description and assessment of the EU ETS, see Kruger and Pizer (2004). An EC Green Paper (2000) also provides some useful background on the design of the system

and nontrading sectors. Second, it specifies how the permits in the trading sector will be distributed among the individual sources. The decision about how much of the target to allocate to the trading sectors also determines residually the stringency of a country's emissions controls on its nontrading sectors.

The EU ETS is being introduced in phases. The first phase (2005–2007) is a kind of “warm-up” phase, during which there is an opportunity to develop experience with the program and see how it needs to be modified in later periods. The second phase (2008–2012) coincides with the period when the EU must meet its Kyoto commitment. The EU then envisions subsequent 5-year (or possibly longer) phases.

Economic Implications of Decentralization

To understand the essential character of the EU ETS and its economic implications, we must first determine the source of demand and supply in the allowance market. In our benchmark case, the central authority determines both demand and supply by specifying the participants in the market and setting the overall cap on emissions. Under the EU ETS, the central authority—the EC—still determines the demand for allowances by specifying the sectors that will participate in the market (although countries can opt to include additional sectors). By knowing who is in the market, we can directly determine, in principle, the demand curve for allowances in the EU. The aggregate demand curve for allowances is again simply the horizontal sum of the demand curves of all the sources in the trading sectors across the member states.

The determination of supply in the trading sector, however, differs from the benchmark case. The member states individually determine what fraction of their national emissions budget they will allocate to the trading sectors. Thus, each country is effectively creating a certain number of allowances, and the aggregate supply of allowances is the sum of these allocations over all the member states. This results in a rather curious system of tradable emissions permits in which the demand curve is centrally determined at the EU level, but the supply curve is determined jointly by the decisions of the member states.⁵ A number of other typically centralized choices have also been decentralized in the EU ETS.

Trading vs. Nontrading Sectors

This structure of decentralized supply and centralized demand decisions has two implications that are worth noting. First, it is difficult for any member state to predict the market price of allowances as they set their own NAP, since one would have to know all the other NAPs in advance. For example, were the other member states to devote a relatively small share of their national allowances to the trading sector, the supply of allowances in the market would be comparatively low and their price high, regardless of what one particular member state chose to do. In turn, this uncertainty about the market price of allowances means that member states will have a hard time in efficiently balancing the level of effort between their trading sectors and nontrading sources. While each member state very clearly sets

⁵The EC does have some indirect control over supply through its approval of member state NAPs, as well as decisions about the use of offsets.

the level of effort required by nontrading sources (by default, the difference between their Kyoto target and their allocation to the trading sectors), the actual effort and emissions from a country's trading sector will depend upon its behavior in the allowance market (i.e., the number of allowances it buys and sells). This leads to the basic conclusion that allowing a decentralized division of allowances between the trading and nontrading sectors will not, in general, result in a cost-minimizing pattern of emissions between trading and nontrading sectors because member states do not possess the information necessary to predict the market price of allowances and set the nontrading sources' level of effort accordingly.

This uncertainty about the market price of allowances could be resolved by centralizing at the EU level the division between trading and nontrading sectors for each member state, leaving the member state to allocate its budget among the respective sources within its borders. This would create a system similar in spirit to the nitrogen oxide (NO_x) trading program in the United States in which each participating state distributes its fixed NO_x "budget" among the sources within that state (Burtraw and Evans 2004). Is this a better idea from an efficiency perspective? The answer is not clear: The centralized EU authority would need to know the costs of each member state's nontrading sources in order to divide efficiently the national target between trading and nontrading sectors in each country. Yet, it is unlikely that the centralized EU authority could assemble such information. This is really part of a more general problem that arises because complete information about costs in all countries and all sectors is unlikely to exist with a single decision-making authority. That makes it hard to reach any general conclusion as to whether it is best (from an economic efficiency perspective) to delegate the decision on dividing allowances between trading and nontrading sectors to the member states or to keep it centralized (Bohringer and Lange 2005). Specifically, what is required is a general equilibrium analysis to compare the efficiency properties of alternative regimes (Parry and Williams 1999).

Equity and Fairness

From a political-economy perspective, the rationale for letting member states make these allocation decisions seems to stem largely from the widespread concern in Europe about "competitiveness" and unfair subsidies being given to particular sources or groups of sources. But does it matter if sources in some countries receive fewer allowances than identical sources in other countries? Since the free distribution of allowances is a one-time wealth transfer to sources, it does raise concerns about equity and fairness. Although in principle the allocation of allowances to a source has no impact on its abatement or production decisions, it can affect its liquidity, including, for example, the need to resort to capital markets for funds. Moreover, there can be some impact on output and prices in regulated sectors. However, over the longer haul, these initial wealth transfers will diminish in significance and may have little impact on the profitability of sources. But regardless of the true impact, this issue looms large in European thinking. Indeed, the EC is required to review and approve each country's NAP, in part to ensure that there are no elements of unfair competition or subsidies (though, in practice, competitiveness does not appear to have played a large role in the EC's criticisms of Phase I NAPs).

Offset Provisions

The EU ETS also offers the possibility of effectively expanding the supply of allowances by obtaining offsets for some EU emissions from outside Europe. More specifically, the EU ETS allows the use of Joint Implementation (JI) and the Clean Development Mechanism (CDM), the same offset provisions available to member states under the Kyoto Protocol.⁶ This holds out the possibility of equalizing marginal costs across trading and nontrading sources if the same pool of offsets feeds both demands. It already seems likely that offsets will play an important role for the nontrading sectors, as there is a widespread recognition that member states have been overly generous in their allocations of allowances to sources in their trading sectors. Without offsets, this generous allocation to the trading sectors implies that very stringent controls will be required on sources in the nontrading sectors if member states are to meet their Kyoto commitments. In fact, it may be that the only way the EU will be able to meet its Kyoto cap is by acquiring emissions offsets through the CDM or JI mechanisms, or through purchasing excess allowances from Russia or Ukraine. But a heavy reliance on offsets from sources outside Europe raises some serious political issues, especially since it is seen by many as being contrary to the spirit of the EU's leadership in the global effort to mitigate climate change.

Auctioning, Banking, and Compliance Provisions

In addition to decentralizing the choice of cap and allocation, the EU ETS gives member states a number of other important responsibilities. Although the bulk of allowances are to be distributed free of charge, there is a provision in the EU ETS that allows individual member states to auction up to 5 percent of their allowances in the first phase and up to 10 percent in the second phase. And, while there is a common requirement that banking be allowed after 2008, member states are free to choose whether and how to allow banking between the 2005–2007 period and the 2008–2012 period. Finally, member states are given considerable latitude to establish domestic compliance procedures, including monitoring, reporting and verification—procedures that are increasingly important as the EU is expanded to countries with weaker institutions and different traditions. Member states have no latitude to introduce something like a safety valve; instead, the EC provides stiff penalties for noncompliance.

The economics literature has made a strong efficiency argument in favor of banking (noted above) and auctioning rather than a free distribution of permits.⁷ However, the potential role for both mechanisms in individual member states may be small, either because decentralization creates fears of placing one country's industries at a competitive

⁶A detailed discussion of the CDM's history, status and prospects is presented in Lecocq and Ambrosi (2007), which appears in this volume.

⁷There exists a large literature that explores the efficiency implications of different forms of market incentives in a setting with preexisting distortions in the economy. Parry, Williams, and Goulder (1999), for example, find that auctions of tradeable CO₂ permits promise significantly larger efficiency gains than programs which grandfather permits, largely because the revenues from the auctions can be used to reduce rates on existing distorting taxes. For a large collection of articles on this issue of "the double dividend," see Goulder (2002).

disadvantage relative to industry in other member states or because of more general distributional concerns. In the case of banking, a member state that allows banking will have a lower cap in the 2008–2012 period because any banked allowances must be subtracted from that country's 2008–2012 Kyoto target. As with an auction, this reduces the pool of free allowances, potentially harming that country's industry relative to industry in member states without auctions or banking. Thus the very freedom given to each member state with regard to auctions and banking may actually discourage their use. On the other hand, there may simply be pressure to maintain greater flexibility for free allocations, regardless of decentralization.

A Discussion of Linking Issues

With the growing number of emissions trading systems at national, regional, and even corporate levels, there has been increased interest in the feasibility of linking distinct programs. This section provides a brief overview of the implementation issues associated with linking issues, describes the basic analytics of linking, and discusses whether linking separate domestic emissions trading programs is always desirable.

Overview of Linking Issues

In theory, linking distinct emissions trading systems will increase efficiency by taking advantage of diverse marginal abatement costs of firms in the larger linked system. In practice, however, there are implementation challenges associated with linking. Several studies have examined the extent to which there is a need for consistency in some of the key design elements of trading programs in linked systems, particularly concerning monitoring and enforcement, allowance distribution, and target type and stringency (for example, Haites and Mullins 2001; Blythe and Bossi 2004; Baron and Philibert 2005 and Baron and Bygrave 2002). Overall, most studies have concluded that it is technically feasible to link different trading systems, but that reconciling differences in design elements may require additional administrative procedures, which may increase administrative costs and complexity.

There are also potential fairness issues raised by linking programs, and these may be more difficult to resolve. Haites and Mullins (2001) note that some participants in a linked system could be worse off than they are in separate systems. For example, when two programs with emission targets of different stringency are linked, prices in the combined program will be higher for one of the programs and lower for the other than they would have been without linking. Victor, House, and Joy (2005) and McKibbin et al. (1999) also note that there may be large capital flows associated with emissions trading when countries with different greenhouse gas obligations are linked. These transfers may be politically controversial if it is perceived that countries are shouldering different burdens for emissions reductions.

Basic Analytics of Linking

We turn now to a discussion of the basic analytics of linking to illustrate what actually happens when countries link their emissions trading programs. In particular, we look at the

effects of linking when countries have different permit prices before linking or when there are differences in the basic architecture of their prelinking trading systems.⁸

Effects of Linking with Differences in Prelinking Prices

Let us start with the question of what happens when two countries with significant differences in their prelinking permit prices link their trading systems. The simple answer is that the price is equalized. Sources in the country with the higher permit price purchase permits from sources in the country where the price of permits is lower until the prices (and hence MACs) are equalized across the two countries with total emissions (i.e., the sum of emissions in the two countries) remaining the same.⁹

While this results in a reduction in overall abatement costs, it does not mean that everyone is better off (Haites and Mullins 2001). Specifically, buyers in the previously low-price country find that they must pay more for additional allowances, while sellers in the higher price country are paid less. But this is really no different from the basic process of opening up trade more generally. The introduction of international trade typically involves winners and losers as a result of the distributional effects from changing prices.

Linking in the Presence of a Safety Valve and/or Banking

Earlier we described two mechanisms for introducing some cost-saving flexibility into the temporal pattern of emission: a safety valve and banking. Banking tends to put a lower bound on prices in each period, as sources see future value in saved allowances and will therefore choose to hold on to them rather than selling them at an unusually low price or using them when relatively inexpensive abatement options remain available. In contrast, a safety valve defines an upper bound on prices in each period that is equal to the specified trigger price. We can see in Figure 1 the effects of these mechanisms on the supply of allowances. In the absence of these mechanisms, the supply of allowances is simply equal to the number of permits issued; in Figure 1, this is represented by a vertical line at the indicated cap level. In an autarkic (i.e., nonlinked) setting of a single country, the introduction of banking and a safety valve would tend to create a price floor at P_e (equal to the expected value of banked allowances) and a price ceiling at P_t , the trigger price (at which point the supply of allowances becomes infinitely elastic). Thus, the supply of allowances has the shape of a step function P_eABD .

Linking raises the intriguing question of what happens to the supply of allowances in the system as a whole when some countries have these mechanisms in place and others do not. Suppose, for example, that country B (without banking or a safety valve) links its trading system to that of country A (which has both banking and a safety valve). In this case, country A will effectively export its banking and safety valve mechanisms to the new consolidated

⁸Additional information on linking systems with different architectures can be found in Marcu and Pizer (2003).

⁹The assumption that total emissions remain the same with linking applies only to emissions covered by the trading program. Emissions outside the trading program, either in nontraded sectors within countries A and B, or in other countries, could change in ways that raise or lower global emissions.

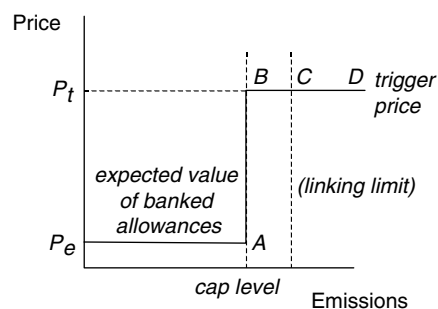


Figure 1. Effect of banking and a safety valve on supply of allowances.

permit market. If the common permit price tips above the trigger price in A, sources in A would pay the trigger price to the government of A for permits that they could then sell at a profit to sources in B. Conversely, if the common permit price dips below the expected value of banked allowances, sources in A will tend to buy permits from sources in B and bank them for future use.

However, it is important to recognize that there may be limits to the exporting of a safety valve, depending on its implementation. Assuming the firms in country A are given a choice between turning in an allowance or paying the trigger price, the exporting process must cease when sources in A have sold all their permits to sources in B and have nothing left to sell. This implies that the potential increase in the supply of allowances for the trading system outside of country A is limited to the supply of permits in country A. Now using Figure 1 to describe the shape of the supply schedule for the linked countries beyond A, the horizontal stretch of the supply curve of allowances at the trigger price can only equal the length of segment *BC* (the supply of allowances in A), at which point the supply curve again becomes vertical (the vertical line at point C). In the EU context, for example, this suggests that if Luxembourg were to have a safety valve, it would only have a very small effect on the potential supply of allowances in the EU ETS outside of Luxembourg (i.e., the horizontal segment *BC* in Figure 1 would be quite short). In contrast, if Germany were to have a safety valve, the increase in the EU supply of allowances at the trigger price would be quite substantial.

A further important consideration is how linking will affect overall emissions now and in the future. If linking changes the use of a safety valve mechanism, this will tend to have a lasting effect on overall emissions, with the direction and magnitude depending on how much the use of the safety valve increases or decreases. Any change will also affect the revenue accruing to the government of country A from payments of the trigger price. In contrast, changes in the use of a banking mechanism will affect current period emission but the change will be offset in a future period.

Linking with a Carbon Tax System

While the safety valve highlights one type of an explicitly price-based program, one might also consider the possibility of linking a completely price-based program, such as a carbon tax system, to a trading program. Suppose country A now has a carbon tax rather than a trading system with a safety valve. It is relatively easy to imagine country A stipulating

that firms can present permits purchased from country B in lieu of a tax payment (perhaps offsets are already used in this way in country A). This will work to equalize marginal costs if country A initially has a higher price when linking occurs. Sources in A will buy permits from sources in B until MACs are equalized at the tax level in A. It is also possible for country A to adjust its tax in response to the price level in country B, with the consequence that the overall emissions level will change as country A changes its tax rate (such a proposal was put forward in New Zealand in 2002; see Government of New Zealand 2005). As in the safety valve case, linking will tend to change the revenue accruing to the government with the carbon tax (country A).

However, it is hard to imagine how linking could work to equalize prices if country A initially has a lower price than country B and does not adjust its tax rate. Since a carbon tax does not provide a currency to trade in, it is not clear how firms in country B could use the lower price in country A to any sort of advantage. This contrasts with the safety valve case, where the permits being sold by the government in country A could be exported to firms in country B and used to increase country B's effective cap.

Linking with Indexed Caps

Yet another possibility arises when country A indexes its cap to output and country B does not. Such an indexed system in country A might look like the proposed Canadian Large Final Emitters (LFE) program, where the cap is adjusted based on output growth. It might also look like the lead phase-down in the United States, where a regulation was written in terms of a rate standard (grams of lead per gallon of gasoline), but sources were allowed to trade. As Fischer (2003) points out, if output in country A is affected by the linking of the systems, this will (by definition) affect the aggregate cap level. Generally, linking raises output in both countries because trading increases efficiency and therefore the productive use of resources. This would tend to raise output and therefore the cap in country A. In some situations, however, where the price in country A is initially lower than in country B, the output level could fall and lower the cap in country A. Other than this effect on the aggregate cap, however, the story is similar to the simple linking story.

Whether or Not to Link

We turn now to the question of whether, in theory, it is always a good idea for countries to link emissions trading programs. The economic underpinnings of linking are seemingly unassailable: once targets are chosen in a given set of countries, trading enhances economic efficiency and reduces overall costs. However, we have already noted two possible weaknesses in this argument. First, there is a tacit, mutual acknowledgement of each others' targets in a linked system, an acknowledgement that might not be desired if one country sees another as having too weak a target. Because emissions targets inevitably will be revised over time, countries necessarily have to think about how their decisions now will affect other countries' decisions in the future. From this perspective, choosing to link one's trading program to another country's weaker regime will likely hinge on whether such action will further encourage a weaker regime or, arguably, give one greater leverage to extract more stringent targets in the future.

Second, although when viewed as a whole, trading leads to overall gains, certain groups do lose. Specifically, as emphasized above, buyers in the low-price country and sellers in the high-price country are made worse off when trading between the countries occurs. Like the case of expanding free trade in general, concerns about adverse impacts on particular groups can thwart progress even though the aggregate gains are well understood.

There is yet a third wrinkle in the argument for linking, and it has to do with the possible effects of linking on technology development. When we broaden the scope of discussion from the EU to the entire world, we have to confront the fact that current mitigation is not the goal. Most countries recognize that even as we reduce emissions now, an equally if not more important goal is to encourage technology development that will allow expanded mitigation opportunities in the future. For example, near-term mitigation consists entirely of energy conservation and fuel switching from coal to gas (given that current, zero-emissions sources produce at capacity). Over the medium term, additional, zero-emissions nuclear and renewable power generation can be constructed. But over the longer term, if we want to stabilize concentrations and therefore halt the flow of greenhouse gases into the atmosphere, all countries have to move towards zero-emissions power generation and transportation, a task unlikely to be met by the set of current, commercially available technologies.

Technology development can be approached in a variety of ways, including market mechanisms that put a price on emissions, but also through direct technology policies (Fischer and Newell 2004). In fact, because of dual market failures—the failure of firms to internalize the externality of pollution damage and the inability of innovators to capture all the rents from their inventions—we actually need two policies to achieve a first-best solution. Even without a first-best solution, however, countries are likely to use different approaches to the dual challenge of mitigation now and technology development for the future. Some countries may seek a high emissions price partly to spur technology development; other countries may prefer to pursue lower prices until more technologies are available, leaning more on direct technology incentives.

The bottom line is that a country seeking to maintain a high emissions price to encourage new technologies may not be happy about a linking mechanism that lowers that price. The problem in this case is that emissions trading is really serving two purposes—mitigation and providing incentives for technology development. Achieving one more cheaply at the expense of the other may not be desirable. These divergent goals are somewhat easy to reconcile within a region, such as the EU, but may be more difficult to reconcile across regions, such as between the United States and the EU, where the balance of mitigation and technology policy has differed. Recent proposals in the United States, for example, have emphasized a linked program where a trading system with a modest emissions price finances a more aggressive technology policy (NCEP 2004; U.S. Senate 2005).

The EU ETS Experience with Linking

This section discusses how the linking issues discussed in the previous section have played out in the specific case of the EU ETS. We examine how variations or inconsistencies in key design elements of member states' domestic programs, such as provisions for emissions monitoring, reporting and verification (MRV), allowance distribution, auctions

and banking, and stringency of emissions targets have affected linking under the EU ETS. We also describe potential and actual steps being taken to address these issues.

Monitoring, Reporting, and Verification Provisions

The MRV provisions in the EU ETS give considerable flexibility to both installations and to member states. There are different “tiers” of methodologies that have different degrees of assumed accuracy. Firms propose installation-specific methodologies to the relevant authority in each member state. Installations are assumed to use the top tiers, but they may petition to use lower tiered methods with lower assumed accuracy if they show that a methodology is impractical or cannot be achieved at reasonable cost. Each member state has the autonomy to grant waivers from use of the top tier methods (European Commission 2004). Member State authorities may require companies to use third-party verifiers if the government does not have the capacity to verify hundreds of emissions reports, and each member state has the authority to set up its own certification procedure for qualified verifiers. It is worth noting that CO₂ monitoring guidelines for most sources in the EU system are relatively straightforward since emissions estimates are based on fuel use, rather than continuous emissions monitors (CEMs), as is done under the U.S. SO₂ and NO_x trading programs.

There has yet to be a thorough evaluation of the degree of consistency among the MRV approaches being used in individual member states. However, early analysis shows that there could be several important differences, including variation in inspection frequency and procedures, differences in overall enforcement rigor, and inconsistent application of the “tiers” or other aspects of the EC’s monitoring guidance (Kruger and Engenhoffer 2006). In addition, there may be differences in the stringency of accreditation procedures for verifiers.

In the short run, member state discretion on the interpretation of monitoring guidelines and the certification of third-party verifiers may undermine some of the consistency that is necessary for an effective monitoring and compliance regime. On the other hand, there are a number of activities underway, both by member states as well as trade associations, to harmonize member state procedures and to encourage common accreditation standards for third-party verifiers (Kruger and Engenhofer 2006). For example, one effort would develop standardized electronic emissions reporting protocols (Kruijd 2006). Furthermore, initial variation in the application of these guidelines will likely diminish over time.

Perhaps more serious are the broader differences in legal systems, enforcement cultures, and administrative capabilities among the twenty-five member states. Will civil and criminal penalties be enforced consistently? Are adequate resources devoted to compliance and enforcement aspects of the program? An uneven approach to enforcement among member states could create unfair competitive advantages for firms in member states with weaker enforcement regimes. Clearly, there are no quick solutions to these issues, as they mirror the broader variation in regulatory institutions and practices throughout the EU.

Allowance Distribution Rules

Several analyses have documented significant differences in the allowance allocation methodologies of member states (Betz, Eichhammer, and Schleich 2004; Zetterberg et al.

2004; DEHSt 2005). For example, some studies have looked at the impact on innovation and investment incentives of different aspects of allocation rules (Mattes et al. 2005; Schleich and Betz 2005) and have found that these rules can affect technology choices and investment decisions. Ahman et al. (2006) and Betz, Eichhammer, and Schleich (2004) examine the impacts of different facility closure and new entrant policies. They find that when member states have policies that require confiscation of allowances after facilities close, they effectively create a subsidy for continued operation of older facilities and therefore a disincentive to build new, cleaner facilities.¹⁰ They further show that different formulas for new entrants can create different incentives for investments across member states. Ellerman (forthcoming) notes that although these provisions increase output and reduce output prices, their impact on allowance prices is ambiguous.

There are a number of proposals to address some of the inconsistencies in allocation methods. For example, some have advocated “benchmarking,” whereby allowances would be allocated according to performance standards that are based on emissions per unit of production (Mistra 2005). Of course, choosing EU-wide benchmarks would be extremely difficult and would raise a variety of equity issues. Others have advocated an increased use of auctions across the EU (Hepburn et al. 2006).

As noted earlier, member states may auction up to 5 percent of their allocations in Phase I and up to 10 percent in Phase II. In Phase I, only four EU member states (Denmark 5 percent, Hungary 2.5 percent, Ireland 0.75 percent and Lithuania 1.5 percent) chose to use an auction (Betz, Eichhammer, and Schleich 2004). There has been a modest increase in the number of member states who plan to auction permits in Phase II. An early analysis of 18 Phase II NAPs shows seven of the member states using an auction provision, with the percentage of allowances auctioned ranging from 0.5 percent in Ireland and the Flanders region of Belgium to 7 percent in the U.K. (Rogge, Schleich, and Betz 2006)¹¹ Neither Germany nor France, two of Europe’s largest economies, has chosen to use an auction mechanism. However, as this article goes to press, there are reports that an auction is still under consideration in Germany (Point Carbon 2007).

In the short term, the highly political nature of allowance distribution and the diverse political pressures in different member states make it unlikely that there will be completely uniform allowance distribution methods across Europe. In fact, for the second phase of the EU ETS, the EC has largely avoided any attempt to harmonize sectoral allowance distribution methods, although it has encouraged simpler and more transparent methods (European Commission 2005). However, to the extent that certain allocation formulas are perceived as inequitable or providing subsidies for national industries, there may be an attempt by the EC to impose more uniformity in the long term.

¹⁰Stavins (2005) finds similar disincentives in vintage differentiated conventional regulations such as the U.S. New Source Review Program.

¹¹The seven member states using an auction in Phase II are Ireland, the U.K., the Flanders region of Belgium, Lithuania, Luxembourg, Poland, and the Netherlands. In addition, Hungary has indicated in a draft NAP that it will auction allowances.

Banking

Prices for a linked system may also be influenced by decisions on banking in individual member states. Although the EU allows each member state to decide whether it will allow banking between the first two periods, most member states ruled out this option. This was largely to avoid building up a volume of banked allowances that would make it more difficult to meet the Kyoto target.¹² Only Poland and France included limited banking provisions in their Phase I NAPs. In theory, the ability to bank allowances through any member state's banking provisions could have had a significant impact on Phase I prices by letting prices rise to reflect future expected prices (Ellerman and Parsons 2006). However, it now appears that the French and Polish provisions will not have a significant impact on Phase I prices because they significantly restrict the number of allowances that may be banked. In November 2006, the EC issued a ruling that any banked Phase I allowances must be deducted entirely from a member state's trading sector NAP, rather than shared between the trading and nontrading sector obligations, further limiting these provisions.

Stringency of Emissions Targets

There has been little analysis of how domestic emissions targets vary in stringency. For example, we know of only one analysis (de Muizon 2006) that estimates the hypothetical autarkic allowance price for each member state based on its allocation to the trading sector. However, there have been analyses that compare targets selected by member states in various other ways. Baron and Philibert (2005) find large differences in volumes allocated to identical sectors across Europe, with allocated emissions for the electricity sector ranging from 30.9 percent above the baseline emissions period used for allocation in Finland to 21.5 percent below the baseline emissions period in the United Kingdom.¹³ Others have found variations among member states in their overall reductions from business-as-usual (BAU) emissions. For example, Grubb, Azar, and Persson (2005) found that emissions targets proposed for the trading sectors in Phase I NAPs varied widely, with a few member states attempting absolute reductions, most proposing growth targets just below a hypothetical BAU projection, and a few with targets above BAU projections.

Such variations in targets among member states are not surprising, nor even necessarily undesirable. Member states face different marginal costs, both inside and outside the trading sectors, which could lead them to set different targets. Moreover, the EU ETS targets are required to reflect the overall national Kyoto (EU burden-sharing) targets, and these targets differ substantially by country in their relation to BAU projections.

Perhaps more problematic is that a lack of consistency in target setting assumptions and methodologies has made it difficult to evaluate the adequacy of targets. The emissions projections contained in member states' NAPs have been developed using a variety of techniques and assumption (Grubb and Neuhoﬀ forthcoming). The EC can require

¹²There is no mechanism under the Kyoto Protocol to increase a country's 2008–2012 emissions target in exchange for early reductions prior to 2008, which is what national governments in the EU would be doing if they allowed banking.

¹³Note that allocation baseline periods may include different years in different member states.

member states to reduce their targets in the trading sector, but there has been a lack of reliable information for evaluating the stringency of a target. The EC has tried to address inconsistent assumptions and projection techniques in its revised guidance for Phase II NAPs (European Commission 2005).¹⁴

Uncertainty about the actual stringency of member state targets has also affected allowance prices, as became clear at the end of the first compliance year of the EU ETS, when the allowance market was surprised by lower than expected emissions and the resulting surpluses of allowances in many member states.¹⁵ As individual countries began to report compliance results and verified emissions were lower than expected, the market price of allowances dropped dramatically. Ultimately, six countries, Austria, Ireland, Spain, the United Kingdom, Italy and Greece, had annual emissions greater than annual allocations.¹⁶ Ellerman and Buchner (2006) note that although there was likely some over-allocation during Phase I of the EU ETS, even greater over-allocation was avoided by reductions in Phase I caps imposed by the European Commission on a number of member states. Without these cuts, the amount of over-allocation might have been almost twice as large.

The price volatility in the EU ETS also illustrates a fundamental risk of a linked emissions trading system. Put simply, decisions about the stringency of emissions targets in one country can affect the allowance prices faced in other countries. In addition to the sharp price drop that occurred when 2005 emissions were announced, there were other, though much less severe, examples of this phenomenon. For example, when Germany submitted its proposed NAP to the European Commission at the end of March 2004, allowances prices dropped. This was reportedly due to a larger-than-expected German allocation (Point Carbon 2004). Similarly, an announcement that the total allocation in the proposed German Phase II NAP would be reduced by 17 million tons led to an increase in the Phase II allowance price in November 2006 (Point Carbon 2006a). Pooling these risks across a larger market can, in general, be a good thing. However, an individual country's perspective depends on whether it perceives itself to be on the giving or receiving side of these events, and particularly whether events turn out to be more strategic than random.

The variation in stringency of member state targets has also had an impact on capital flows between firms in different member states and raised concerns about inequitable sharing of the burden of emissions reductions. Not surprisingly, some of the most negative reaction to the first year of the EU ETS has come from some sources in the United Kingdom, which had the most stringent target in the EU ETS and was a large net purchaser of emissions allowances. One study estimates that in 2005, U.K. companies had to purchase allowances from companies in other member states at a cost of £ 475M (Open Europe 2006). On the other hand, the ability to borrow within a compliance period (e.g., from 2006 to 2005) has attenuated some of these effects (Point Carbon 2006b). Thus far, the issue of capital

¹⁴Despite efforts to promote consistency, a recent analysis of Phase II NAPs finds that assumptions, data quality, and methodologies continue to differ in member state projections of BAU. See Rathmann et al. 2006.

¹⁵A plausible alternative explanation is offered by Ellerman and Buchner (2006), who note that lower than expected emissions may have been the result of greater than expected abatement.

¹⁶Note that estimates of allocations vs. reported emissions are preliminary and do not necessarily include allocations from new entrant reserves (see Ellerman and Buchner 2006).

flows between member states has not been a major issue in Europe, and the levels of capital flows involved have been small. However, the issue could grow in importance if there is a perception in subsequent phases of the EU ETS that variation in target stringency is accentuating winners and losers at the member state or firm level.

Ultimately, the European Commission may decide to exert more control over individual member state targets in order to enforce more uniformity of effort and to support a price that is consistent with EU-wide policy. In fact, as this article goes to press, the Commission is ordering even more aggressive cuts in member state caps for Phase II than it did for Phase I. More generally, concerns about inconsistencies between member state programs could lead to an increasingly stronger role for the Commission and a somewhat more centralized EU ETS.¹⁷ However, it is unclear whether, in the long term, this method of reducing individual member state targets will be considered a transparent, effective and equitable approach for imposing the desired level of target stringency.

Implications of the EU ETS for Global Linking

The recent experience with the EU ETS provides a useful lens through which to look at the issue of establishing a broader international emissions trading regime. We would expect all of the EU ETS experience described above concerning heterogeneous MRV, allowance distribution, stringency of emissions targets, and architecture to apply also at the global level. However, at a global level, the variation in enforcement capacity, allocation choices, and stringency will be even greater than within the EU ETS, only without the cohesion and authority of the European Commission.

Variations in Institutional Capacity for MRV

The EU ETS experience is particularly relevant regarding divergent institutions for measuring, reporting, and verification. The issue faced by the EU ETS concerning differences in cultures of enforcement and administrative capacity among member states is only a fraction of what might exist if one were to compare the EU as a whole to, say, Russia or China. While this is less likely to be an issue among countries or regions such as the EU, Japan, and the United States, by most accounts, it is emissions trading with developing countries that is necessary to lower costs substantially. Studies of the Kyoto Protocol, for example, found the most dramatic effect in lowering costs to occur when trading was expanded to include developing countries (Weyant and Hill 1999). Concerns about institutional capacity have been raised for some time, even as international climate negotiations and scholarship have focused on the idea that developing countries and others with potentially weaker legal and economic institutions ought to embrace market mechanisms (Bell and Russell 2002). If it turns out that institutions in these countries cannot support emissions trading, it will be important to consider other, presumably less efficient, mechanisms to get at these cheap mitigation opportunities, including continued project-based approaches such

¹⁷The Commission is also considering a number of steps to make allowance distribution, monitoring and verification, and other aspects of program design more harmonized between member states after 2012. See European Commission (2006).

as the CDM, broader crediting programs for sectoral efforts or regulatory reform (such as efficiency standards or removing harmful subsidies), and government-to-government efforts on major energy deals.

Variation in Allowance Distribution

The variation in approaches to allowance allocation is also likely to be larger when we look outside the EU. Proposals in the United States, for example, have suggested larger auctions (RGGI 2005) and allocation to businesses at different points in the fossil-fuel chain (U.S. Senate 2005). However, these differences are probably less important in a discussion of linking and decentralization at the global level. Part of the reason allocation is high on the list of concerns within the EU is that the EU is a single market with fluid factors of production and movement of goods. The United States often faces the same pressures with regard to harmonization of environmental regulations and other rules across states (Oates 2002). At the global level, however, the existing differences among regulations and laws in different regions, as well as obstacles to the fluid movement of capital and labor, mean that there is likely to be less pressure concerning allowance allocation. Allocation concerns that figure prominently in the EU, such as new entrant allocations and effects on new plant locations, are simply dwarfed at the global level, where there are more major concerns regarding plant location across regions rather than within.

The single area where allocation might draw significant attention on the global level would be if such choices had an impact on the price of traded goods (and therefore existing producers in other countries). As noted earlier, allocation generally does not affect output prices unless output markets are regulated (e.g., electricity) or unless the allocation is updated in response to changes in output or input. If some countries choose to use updated allocations in sectors with tradable goods, other countries might see this as an unfair trade practice. However, this concern would exist irrespective of linking or coordination of climate policy.

Variation in Stringency and Architecture of Emissions Targets

Perhaps the most significant concern about the EU ETS that would be relevant at the global level is the variation in the levels of stringency and architecture of emissions targets in different regions. There are at least two reasons why there might be a problem with linking programs with different levels of stringency (Pizer 2006). First, as noted in our above discussion about the EU ETS, the flow of allowances among countries (e.g., into the UK), coupled with the existing perception that some countries have adopted more aggressive targets in their NAPs, has made some member states question the fairness of NAPs. Second, there is a concern that capital flows associated with large net sales of allowances across borders will itself be an adverse consequence.

Imagine, now, the much greater possible variation in stringency that is likely to exist at the global level. Developing countries, were they ever to create trading programs and join an international regime, would likely start out near their BAU levels (Frankel 2006). Such approaches are arguably fair, based on a variety of notions of national responsibility, but will they hold up in light of large trade flows? Perhaps of more concern would be programs in other industrialized countries, like the United States, that might be less stringent (measured

by the allowance price prior to linking) relative to the EU. Would Europe open the EU ETS to a U.S. market where the net effect would be that Europeans pay U.S. companies for allowances whose relative abundance in the United States was a political decision? Would such a linkage change the dynamics for future target-setting in these countries? All of these concerns are only amplified when national programs use different architectures, such as indexed targets or a safety valve. It seems virtually impossible to imagine a country without a safety valve agreeing to link to a country with a safety valve with a low trigger price, as emissions will likely rise with a consequent increase in revenue to the government with the safety valve.

A related concern is whether the designers of national emissions trading programs have tried to set the price in their programs (explicitly or implicitly) to meet certain domestic needs and/or constraints, versus setting the cap without regard to price. Because linking programs means equalizing permit prices, the new price might not meet those needs or constraints. What might those needs or constraints be? One is the distribution of costs within a country. Even if linking makes a country better off as a whole, some may lose. Another issue arises if one country is seeking higher permit prices to drive technology development, and linking lowers the price.

Conclusions

The EU ETS essentially links the domestic trading programs of 25 countries, which, although subject to some common standards and oversight from the EC, still have considerable autonomy. This experiment with linked systems provides a useful laboratory for considering the political, economic and administrative challenges that would be faced by a global trading system, which will likely be even more decentralized than the EU ETS but with less oversight. An important, early observation is that linking makes it increasingly difficult to achieve an efficient balance of the burden across trading and nontrading sources (though access of both sources to a common offset market may alleviate some of this difficulty).

The current Kyoto Protocol has the appearance of something similar to the EU ETS, with provisions of the Protocol resulting in some degree of harmonization over national and (in the case of the EU) regional policies and the ability of governments to trade national commitments. For example, parties to the Kyoto Protocol have a centralized system for approving project-based credits in developing countries through the CDM Executive Board and common reporting standards.

Yet, looking at the policies pursued by different Kyoto signatories—namely the absence of any national trading programs outside of Europe—it is clear that the degree of supranational authority is much lower than in the EU ETS. Moreover, imagining a future regime that includes the United States suggests an international system with even less centralized authority than Kyoto.

In sum, the model of decentralization in the EU ETS has broken new ground in our experience with emissions trading regimes across multiple jurisdictions. It is providing new evidence on how different approaches to enforcement and monitoring, allocation, and even effort and stringency, can be encompassed in a single trading program. This experience is particularly valuable as we think about how a global regime might evolve.

The differences between the EU experience and the global context suggest that the challenges of a global system are likely to be even more formidable. On the enforcement

and institution side, this suggests that broad-based emissions trading within developing countries may not be a realistic goal in the near term, and other avenues for engagement and trade need to be explored. Allocation is likely to be less of an issue, however, because the mobility of capital and labor is lower globally than within the EU. On the other hand, concerns over different choices about target stringency and effort are likely to loom large. Increasing efforts by the EC to tighten and effectively centralize member states' allocations has no global analogy. A global regime will also have to confront differences in architecture. The EU ETS has followed the traditional approach of an absolute cap. But other countries have discussed the possibility of price-based mechanisms and indexed caps, as well as a wide range of nonmarket mechanisms (e.g., standards and technology mandates). All of this may mean that in the short term, other national programs will not link to the EU ETS (assuming mandatory programs arise in other nations).

This raises the possibility of an alternative to linking: price harmonization. Countries could set their domestic policies in ways that recognize and respond to the efforts in other countries in an effort to harmonize marginal costs. New Zealand showed some interest in this approach when it proposed a carbon tax linked in value to the average permit price in the EU ETS. It is also somewhat remarkable that other domestic proposals suggest similar price levels. For example, the trading price in the EU ETS has been around €15 (\$18) per ton of CO₂, while the proposed safety valve in Canada was C\$15 (\$13). In one U.S. proposal the projected price was \$7, and the level of a proposed Japanese tax was ¥2500–3000 (\$6–7). While not identical, these prices are much closer to each other than estimates of autarkic prices in response to the Kyoto Protocol. The experience in Europe, as well as signals about evolving policies in other countries, suggests that a natural tendency towards price harmonization may be inevitable.

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Policy Monitor**Edited by Maureen Cropper****The Clean Development Mechanism:
History, Status, and Prospects****Franck Lecocq* and Philippe Ambrosi******Introduction**

The Clean Development Mechanism, or CDM, was a late invention in the negotiation of the Kyoto Protocol—so late, in fact, that it has been called the “Kyoto surprise” (Werksman 1998). In June 1997, only six months before the Kyoto negotiations, the Brazilian delegation proposed to create a Green Development Fund (GDF) that would be supported by countries out of compliance with their commitments, and that would support mitigation projects in developing countries. Though endorsed by the G77 and China, this proposal did not fly because developed countries were strongly opposed to penalties for noncompliance. Developing countries, on the other hand, were strongly opposed to any mechanism that would replicate the logic of the Activities Implemented Jointly (AIJ) of the UN Framework Convention on Climate Change (UNFCCC) (i.e., any mechanism whereby Annex I countries¹ could offset some of their commitments through emission-reducing projects in developing countries).

To reach an agreement, the United States and Brazilian negotiators suggested in November 1997 that the GDF be turned into a “positive” scheme whereby countries with commitments under the Kyoto Protocol would be allowed to exceed their emissions quotas

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¹The Annex I Parties to the UNFCCC are Australia, Austria, Belarus, Belgium, Bulgaria, Canada, Czech Republic, Denmark, Estonia, European Community, Finland, France, Germany, Greece, Hungary, Iceland, Ireland, Italy, Japan, Latvia, Liechtenstein, Lithuania, Luxembourg, Monaco, Netherlands, New Zealand, Norway, Poland, Portugal, Romania, Russian Federation, Slovakia, Slovenia, Spain, Sweden, Switzerland, Turkey, Ukraine, United Kingdom, and the United States of America.

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by supporting emission reduction projects in developing countries. Unlike AIJ, however, the new mechanism would put as much emphasis on “promot[ing] sustainable development” as on “help[ing] developed countries meet their commitments.” After intense negotiations, the CDM was finally included as Article 12 of the Kyoto Protocol, signed 11 December 1997.

As often happens in international negotiations, the agreed text left many ambiguities unresolved. First, parties differed in their interpretation of the new emphasis on development. Most developed countries still viewed the CDM as a way to gain access to cheap mitigation opportunities in developing countries, and thus to reduce their mitigation costs. But developing countries were looking at the CDM as a new channel for development assistance (Grubb et al. 1999). These two interpretations of the CDM are not necessarily compatible.

A second glaring difficulty was that CDM projects would create new credits in countries without commitments, and would result in the transfer of those credits to countries with commitments, thereby increasing the total amount of emission credits in circulation. Such “money creation” carries obvious risks that were not lost to the negotiators. As the Chairman of the Kyoto Conference put it, “Though I did facilitate the approval [of the CDM], I did not like it. . . . I do not understand how commitments can be implemented jointly if only one of the Parties involved is committed to limit or reduce emissions.” (Estrada-Oyuela 1998, p. 25).

Yet, despite these fundamental difficulties, and despite the vagaries of the international climate negotiations in the years following the signing of the Kyoto Protocol, in less than a decade the CDM has turned into a vibrant market. In 2005 alone, more than 180 transactions were recorded, channeling \$2.5 billion of carbon finance to developing countries—an amount equal to 2.5 percent of total net Official Development Assistance (ODA) (Capoor and Ambrosi 2006a). The CDM now has a large number of stakeholders, both in the North and in the South. The CDM, furthermore, has been instrumental in the involvement of developing countries in the Kyoto Protocol, an outcome that was not obvious in 1997.

The objective of this article is to review how this remarkable turn of events unfolded, to examine whether and to what extent the CDM has overcome the structural difficulties highlighted above, and to discuss the future of the CDM in the context of global climate mitigation in the medium and long run. The article is structured as follows. It first reviews the history of the CDM from Kyoto to the present day. Then it describes the current status of the CDM. Next it assesses the relationship between the CDM and sustainable development. Finally, it discusses the remaining challenges and future prospects for the CDM.

A Brief History of the CDM

The Kyoto Protocol (KP) (1997) calls for industrialized countries and economies in transition—the so-called Annex B countries²—not to exceed certain greenhouse gas (GHG) emission targets during the first commitment period (2008–2012). In addition to domestic policies and measures, Annex B Parties can meet their targets by using three flexibility mechanisms: (i) purchasing Assigned Amount Units (AAUs)—emissions

²The list of Annex B countries is the same as the list of Annex I countries, except for Turkey, which is in Annex I but not in Annex B because it had not ratified the UNFCCC at the time the KP was signed.

allowances under the KP—from other Annex B Parties under International Emissions Trading (KP Article 17), (ii) contributing to emission-reducing projects in other Annex B Parties and acquiring Emission Reduction Units (ERUs) through Joint Implementation (JI) (KP Article 6), and (iii) contributing to emission-reducing projects in non-Annex B countries through the CDM (KP Article 12).

The Negotiations from Kyoto to Marrakesh (1997–2001)

Though Article 12 sets out important principles, the CDM was little more than an empty shell after Kyoto. The main operational guidelines of the CDM were agreed upon only in November 2001, as part of the Marrakesh Accords. And the process was only complete in 2003 with the agreement over the rules governing forestry-related CDM projects—the so-called Land Use, Land-Use Change and Forestry (LULUCF) projects.

The reasons why it took the international community four years to negotiate the Marrakesh Accords have been amply discussed elsewhere (e.g., Bodansky 2001; Hourcade 2002). They go far beyond the CDM and have to do with the stringency of the Kyoto targets, with the uncertainty over the costs of meeting these targets, and with a series of misunderstandings between the two major negotiating blocks, Europe and the United States. What matters here is that the debates leading to Marrakesh had a lasting influence on the rules and the operations of the CDM.

The first key issue after Kyoto was that the flexibility mechanisms were strongly opposed by many stakeholders, notably a large number of NGOs and some of the Green Parties then in charge of climate negotiations in key European countries. These stakeholders saw flexibility as a way for industrialized countries to escape their obligations to reduce their domestic emissions. Their criticism of the flexibility mechanisms also reflected a broad anti-market rhetoric, particularly prevalent in Europe. Although these critics have achieved little of their policy agenda (e.g., a proposal for quantitative caps on flexibility was finally left out of the Marrakesh Accords), the Kyoto flexibility mechanisms in general, and the CDM in particular, still remain strongly suspect in many quarters.

Second, the CDM turned out to be particularly controversial among the flexibility mechanisms because, as noted above, it creates new emission credits. Article 12.5(c) of the KP states that only emission reductions that are “additional to any that would occur in the absence of the certified project” are admissible. But the counterfactual is by construction impossible to observe, and clearly open to strategic manipulations. Since both the buyer and the seller of emission reductions have an incentive to inflate the baseline, the risks are high that the CDM may open up a major loophole in the Kyoto Protocol. The pressure was thus very strong for the CDM to prove beyond a doubt that it was environmentally additional, which had two major consequences. First, it was decided that additionality would be tested on a project-by-project basis, and not at the program level as some had originally envisioned. Second, the Executive Board (EB) of the CDM, the body designated by the Kyoto Protocol to supervise the CDM, took a very conservative approach to the validation of emission reductions.

Not surprisingly, the balance between the mitigation and development objectives of the CDM was the subject of intense discussions. Of particular importance was the distribution of rents between the North and the South: the risk was that the North would purchase emission reductions cheaply by harvesting ‘low-hanging fruits’ from the South (Hourcade

and Toman 2000). However, specific proposals to regulate rent sharing in CDM projects were rejected. It was agreed that it would be left to the host country to determine whether a particular CDM project is compatible with its sustainable development priorities. As we will see below, however, the debate over the relationship between CDM and sustainable development is far from over.

Finally, LULUCF activities were often seen as particularly doubtful from an environmental standpoint because of measurement uncertainties. The suspicion over LULUCF was compounded by the fact that the scope of forest management under Article 3.4 of the KP was explicitly negotiated among Annex 1 parties as a means to relieve some of the pressure created by the Kyoto targets. LULUCF projects in the CDM faced even higher criticism from some stakeholders, notably environmental NGOs. LULUCF projects, critics argued, would be environmentally unsound, would flood the market with unsound credits, and would lead to environmental catastrophes in the South because they would favor fast-growing industrial plantations of alien species over community-based, sustainable forest management. This pressure led to a strict limitation of the scope of LULUCF projects under the CDM in the Marrakesh Accords, both from a qualitative point of view (only afforestation and reforestation projects are allowed) and from a quantitative point of view (the total amount of LULUCF Certified Emission Reductions [CERs] that can be obtained is capped). NGOs also succeeded in imposing the restriction that credits from LULUCF CDM projects cannot be imported into the EU Emissions Trading Scheme (EU-ETS).

As per the Marrakesh Accords, the CDM project cycle is as follows. First, the project proponent—for example, the project sponsor, one of the investors, the potential carbon buyer, or a third-party (e.g., a consultant company)—produces the Project Design Document or PDD. The PDD includes, *inter alia*, a description of the project, an explanation of how the baseline and monitoring methodology will be applied, a discussion of the environmental impacts of the project, and a compilation of stakeholders' comments, if any. In addition, the buyer(s) and the seller—even if they are private entities—must each get a Letter of Approval (LoA) from the entity in charge of reviewing CDM projects in their respective governments, the Designated National Authority or (DNA). The LoA states that the country approves participation in the project, and for the host country, that the project contributes to sustainable development.

Once finalized, the PDD and the LoAs are validated by an independent third party (typically an auditing company) accredited by the CDM EB—the Designated Operational Entity (DOE). By validating the project, the DOE determines that the project has been approved by the parties involved, and that it correctly applies the selected baseline and monitoring methodology. The DOE then submits the PDD to the CDM EB for registration. (If there is no off-the-shelf baseline and monitoring methodology available, the DOE first submits a new methodology for validation by the EB, and once the methodology is approved, the DOE submits the PDD.)

Finally, once the project is registered and has become operational, a second DOE is charged with reviewing and certifying the emission reductions generated by the project. The CERs are formally issued by the EB and transferred to the project participants' accounts. At that point, CERs are essentially fungible with other Kyoto allowances such as AAUs or ERUs.

The Emergence of a Market (1999–2005)

Interestingly, the CDM market emerged before the rules governing the CDM were finalized. In fact, when Russia agreed to ratify the Protocol in October 2004, thereby making it certain that the Kyoto Protocol would enter into force, more than 120 transactions had already been recorded (Lecocq 2005). Carbon projects had been tested even before the KP was signed, in the context of the Pilot Phase of the AIJ. But AIJ projects did not lead to transfers of credits, and were heavily criticized for poor environmental integrity (Michaelowa 1999). In the late 1990s and early 2000s, American and Canadian companies had also started to undertake carbon projects, often as part of company-wide voluntary commitments to limit greenhouse gas emissions. These projects, however, were not intended to be validated under the CDM and did not follow CDM guidelines.

The participants in the Prototype Carbon Fund (PCF), six governments and fifteen private companies, were the first investors in the CDM. The PCF is a closed \$180 million mutual fund managed by the World Bank to purchase emission reduction credits under JI and the CDM. The PCF was established in 1999, became operational in April 2000, and signed its first emission reduction purchase agreement for a CDM project in Chile in 2002. The motivation of PCF participants included learning about this emerging market, gaining competitive and strategic advantage over competitors, influencing ongoing negotiations (the PCF was explicitly set up as a vehicle for informing negotiators about real-world implementation of CDM projects), and acquiring emission reductions. Although it invested very little of its own resources into the PCF, the World Bank saw carbon finance as an opportunity to channel additional resources, private resources in particular, to developing countries in a period of declining ODA. The PCF is also in line with the Bank tradition of innovation in financial markets.

Another key player in the early market was the Government of the Netherlands, which had decided early on to purchase emission reductions through flexibility mechanisms as part of a comprehensive strategy to meet its Kyoto target. In addition to participating in the PCF, the Government of the Netherlands also developed the first carbon tenders for CDM and JI (2001). In 2004, the two original players in the CDM market—the Government of the Netherlands and the World Bank (whose carbon finance activity had by then grown to include new funds besides the PCF)—still represented about a third of the total volume of project-based transactions (Lecocq 2005).

The adoption of the Marrakesh Accords in December 2001 led more players to move in. Private firms from Japan started to enter the market in 2002 and 2003, despite the absence of a domestic climate policy in Japan (the Japanese climate policy was approved only at the end of 2005). European firms followed about a year later, when it became clear (i) that the EU Emissions Trading Scheme (see the articles in this issue by Convery and Redmond, Ellerman and Buchner, and Kruger, Oates, and Pizer) would become operational, and (ii) that CERs would become eligible at least in part, under the EU-ETS. Among the most recent entrants in the market are Annex B Governments—some of which have earmarked massive amounts of money to purchase CERs—after entry into force of the KP. More recently, a wide range of buyers, such as banks or speculators, that do not need CERs for compliance but aim to trade them on the secondary market, have entered the CDM market. It is estimated that in at least one-third of all the project-based transactions concluded between January 2005

and April 2006, the buyer had the intention of selling some of the resulting CERs on the secondary market (Capoor and Ambrosi 2006a).

The regulatory buildup was slow to catch up with the explosion of the market, as the Executive Board and its technical panels were crippled by lack of resources in the face of a rapidly growing backlog of projects. Tensions over the “regulatory bottlenecks” have subsided somewhat as methodologies have been validated for a large range of project activities, and as regulatory resources have increased. An indicator of this improvement is the fact that, although the backlog of projects to the EB is still growing because of a rapid increase in the supply of projects, the average length of CDM project cycles is decreasing (Fenhann 2006). Overall, the international administrative framework that supports the CDM is now considered to be operational (UNDP 2006). In addition, DNAs have been set up in 112 countries, of which 91 are developing countries, thereby making project approval faster. Moreover, an increasing number of developing countries have set up mechanisms to promote project opportunities to the international CDM market.

The CDM Market Today

The CDM market cannot be understood independently of the broader “carbon market” to which it belongs. The carbon market is defined here as the sum of all transactions in which one or several parties pay another party or set of parties in exchange for a given quantity of “GHG emission credits.” The legal definition of these credits varies, but what is important is that they are transferred from the seller to the buyer. Payments can take various forms, such as cash, equity, debt, or technology transfer.

Carbon transactions can be grouped into two main categories:

1. Allowance-based transactions, in which the buyer purchases emissions allowances created and allocated (or auctioned) by regulators under cap-and-trade regimes, such as Assigned Amount Units (AAUs) under the Kyoto Protocol, or EU Allowances (EUAs) under the EU-ETS.
2. Project-based transactions, in which the buyer purchases emission credits from a project that reduces GHG emissions compared to what would have happened otherwise. Project-based transactions include CDM and JI transactions, but also non-Kyoto transactions such as voluntary transactions in Europe or in the United States (from entities seeking to offset emissions related to, *inter alia*, their business operations, a service they offer or an event), and projects related to non-Kyoto regulations in Australia and in some US States.

Allowance- and project-based transactions differ by the risks attached to them. In an allowance-based transaction, the asset being traded (the allowance) exists before the transaction. The main risk is therefore delivery risk. In a project-based transaction, the asset being traded is created during the process. So in addition to the delivery risk, there is a “noncreation” risk. For example, the project may underperform and not generate the expected amount of emission reductions (project risk), political or institutional problems may occur and put the project and the generation of emission reductions in jeopardy (country risk), or the regulator may refuse to certify the emission reductions (nonregistration risk). Of course, if the transaction is concluded after the issuance of the CERs, the risk of

noncreation no longer exists and the transaction is equivalent, in terms of risk, to an allowance-based transaction. Transactions of issued CERs on the secondary market are becoming more common. At the time of writing, however, secondary transactions are only forward. Spot transactions of issued CERs remain technically impossible because the International Transaction Log—which connects the CDM registry (where CERs are issued) to registries of Annex B countries (where, once transferred, CERs can be used for compliance or further transacted)—is not operational.

A Fast-Growing Market

Total volumes traded on the carbon market, excluding voluntary transactions, reached 717 million metric tons of CO₂ equivalent (MtCO₂e) in 2005, a 5.8-fold increase over 2004, and amounted to 1023 MtCO₂e in the first three quarters of 2006. To put this in perspective, 717 MtCO₂e represents about 6 percent of the total 1990 GHG emissions by KP signatories, or roughly the annual CO₂ emissions of France and Spain combined. Volumes exchanged through project-based transactions and allowance trading were still roughly equal in 2005 (332 MtCO₂e versus 384 MtCO₂e) but allowance trading is now dominant with 74 percent of the volume exchanged in the first three quarters of 2006 (Capoor and Ambrosi 2006a, b).

The rapid growth of the carbon market is a direct consequence of the entry into force of the EU-ETS (January 1, 2005) and the Kyoto Protocol (February 16, 2005). Both boost demand for CDM projects, because CERs can be used to comply with Kyoto obligations and with EU-ETS obligations. As a result, the EU-ETS market also has a strong price effect on the CDM market.

The total volumes traded through project-based transactions in the carbon market are growing rapidly: 24 MtCO₂e in 2002, 51 MtCO₂e in 2003, 110 MtCO₂e in 2004, 384 MtCO₂e in 2005, and 234 MtCO₂e in the first three quarters of 2006 (Lecocq and Capoor 2005; Capoor and Ambrosi 2006a,b). These volumes represent the total amounts of emission reductions that sellers are planning to deliver up to 2012 based on transactions signed during the year in question. The volumes of emission reductions that effectively changed hands each year are much lower. These numbers are conservative because carbon traders are under no obligation to record CDM transactions before registration, hence some transactions may not have been accounted for.

The growth of the CDM market can also be seen in the annual number of projects submitted for validation. This number has grown exponentially from 5 in 2003 to 58 in 2004, 491 in 2005, and 676 in the first three quarters of 2006 (Fenhann 2006). Overall, a total of 386 projects totaling some 660 MtCO₂e have been registered as of the end of October 2006 (<http://cdm.unfccc.int/Statistics>). Yet another indication of the rapid growth of the CDM market is the capitalization of carbon funds worldwide, which has surged from approximately \$275 million in January 2004 to an estimated \$4.6 billion in April 2006 (Bulleid 2006), and to an estimated \$6.4 billion in September 2006 (New Carbon Finance 2006). There is thus no shortage of demand for projects-based emission reductions.

Who Is Buying?

Figure 1 shows the share of market buyers in the total volume of carbon traded from January 2005 to September 2006. In this chart, emission reductions purchased by funds

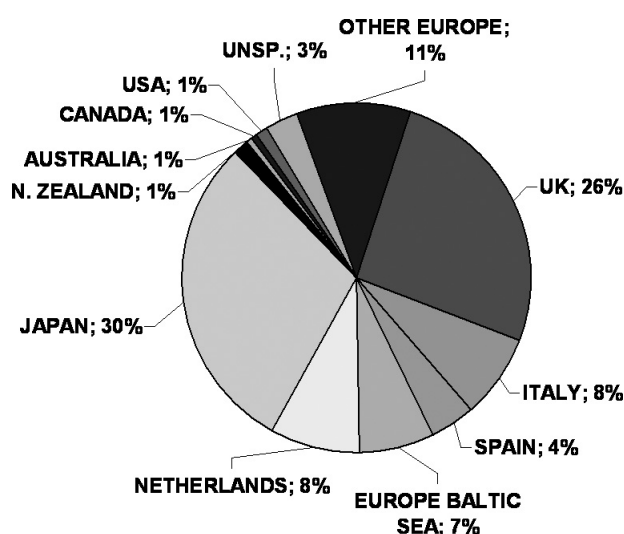


Figure 1. Major buyers in project-based transactions (as a share of total volume exchanged, vintages to 2012) from January 2005 to September 2006. EUROPE-BALTIC SEA refers to Finland, Sweden, Norway, Germany, Denmark, and Iceland. OTHER EUROPE refers to France, Portugal, Switzerland, Austria, Belgium, Luxembourg, and Greece. UNSP. refers to purchases where the origin of buyers could not be verified.

Source: Capoor and Ambrosi, 2006b.

are allocated to fund participants *pro rata* their shares. The market is overwhelmingly dominated by European (64 percent) and Japanese (30 percent) buyers. Overall, the private sector is dominant, with more than 80 percent of the volumes purchased. However, the share of the private sector may diminish in the future as the large commitments recently made by governments result in actual transactions.

Canada's very small share (1 percent) may come as a surprise as Canada's emissions in 2004 were 33 percent above its KP target. The continuing uncertainty over Canadian climate policies may explain this low number. The small volume of purchases from the United States and Australia relate to non-Kyoto projects.

Finally, as noted above, an increasing number of buyers purchase for resale. Japanese firms, but also funds based in the United Kingdom and even the United States, some of which are speculative, have entered this rapidly developing market. In particular, purchases for resale, much more than domestic needs, explain the United Kingdom's large share of purchases.

Who Is Selling?

As Figure 2 shows, China captured nearly two-thirds of the market for project-based transactions from January 2005 to September 2006. Latin America (16 percent) and the rest of Asia (12 percent) account for most of the remainder, well beyond Africa (4 percent). Projects in other regions relate either to JI or to non-Kyoto projects. These aggregates, however, are strongly influenced by the conclusion of a handful of "mega-deals" for trifluoromethane (HFC-23) destruction in China. HF-C23 is a powerful greenhouse gas, one ton of which is equivalent to 11,700 tons of CO₂. When HFC-23 deals are taken out

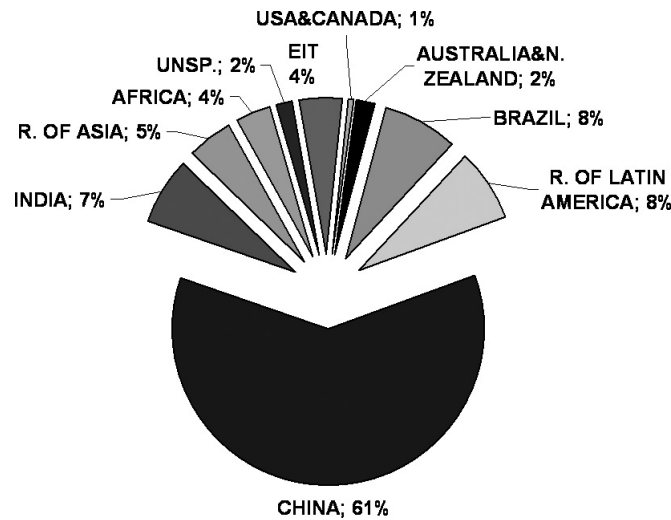


Figure 2. Major sellers in project-based transactions (as a share of total volume exchanged) from January 2005 to September 2006.

Source: Capoor and Ambrosi 2006b.

of the picture, China's share of supply, though still dominant, falls to about 40 percent. This distinction is important because most of the opportunities for HFC-23 destruction worldwide appear to have now been exploited.

With or without HFC-23, the market is concentrated in large countries. Three countries (China, Brazil, India) account for nearly 80 percent of the supply (about half without HFC-23), even though 37 countries have at least one project registered at the time of writing (Fenhann 2006). With about 4 percent, the small share of African countries in total supply is particularly striking. In fact, registered projects exist only in South Africa and in the Maghreb. And apart from South Africa, the Maghreb countries and a few sub-Saharan countries, Africa is essentially absent from the CDM portfolio (Fenhann 2006).

Balance among Technologies

As indicated in Figure 3, HFC-23 destruction projects largely dominated the volumes sold (52 percent) between January 2005 and September 2006. Methane capture from landfill gas (LFG) and coal mines (CMM) form the second largest group (13 percent), and renewable energy projects (wind, hydro, biomass, other renewables) constitute the third (12 percent). Energy efficiency (6 percent), which includes fuel switching, and N₂O destruction (5 percent) follow. Overall, projects abating gases other than CO₂ account for at least 70 percent of the volume. This is a totally unexpected outcome, as most commentators anticipated fuel switching, energy efficiency and LULUCF projects to constitute the bulk of the CDM.

The main reason for this distribution is that, at carbon prices below \$5/tCO₂e, carbon revenues have only a marginal impact on the internal rate of return of energy efficiency or renewable energy projects (Bishop and Lecocq 2004). Even with the higher carbon prices observed now, the carbon revenue effect is diluted by the fact that these projects typically require 5 to 7 years before they can be implemented, thus limiting the volume of credits

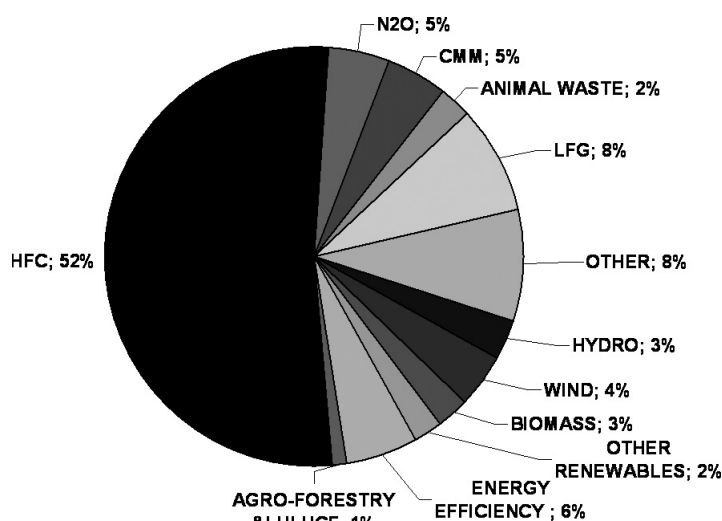


Figure 3. Distribution of technologies in project-based transactions (as a share of total volume supplied) from January 2005 to September 2006.

Source: Capoor and Ambrosi 2006a, b.

they can generate before 2012 and, consequently, the amount of carbon revenues they can obtain. Noncarbon projects, on the other hand, benefit from a much higher price per metric ton of gas avoided because of the conversion factors used in the Kyoto Protocol (e.g., HFC-23 is 11,700 times as potent as CO₂, and is thus worth 11,700 times as much), shorter construction periods before implementation, lower upfront investment, and lower performance risk (especially for the early vintages that are most valued by buyers).

Structure of Contracts and Price of Carbon

In line with the development objective of the CDM, it was anticipated that the CDM would essentially follow an investment model, with the developed country entity investing equity or debt in the project and receiving emission reductions (instead of or in addition to returns and debt service) in exchange. But an alternative commodity model has emerged whereby the buyer simply purchases the emission reductions produced by the project without providing equity or debt.

The commodity model is now dominant, mostly because the skills required to purchase carbon (evaluating baselines, etc.) are different from the skills required to invest in projects. Also, firms that invest in projects in developing countries do not necessarily need credits, or if they do, do not necessarily invest in the classes of projects that produce emission reductions at the lowest cost. Conversely, carbon buyers are in most cases not in the business of investing in developing countries. As a result, carbon contracts today are usually akin to a power purchase agreement in the energy sector, with strong implications for financing (see next section).

Within these broad parameters, the structure of CDM contracts has evolved very rapidly over the past year. Between the end of 2004 and early 2005, most transactions were forward purchases with payment on delivery at a fixed price per ton of CO₂e, over a period of a few to ten years (and usually no later than 2012). The key differentiation across contracts was the

sharing of the nonregistration risk. Some buyers were ready to pay for emission reductions as soon as they were verified by a DOE, whereas other buyers insisted on getting *certified* ERs, with punitive provisions (e.g., fines) in case of default. The price per ton was naturally higher when the registration risk was borne by the seller (Lecocq and Capoor 2005).

As of late 2006, the fixed-priced forward contract is no longer the dominant contract on the market. A flurry of new contractual structures has been introduced, with no clear standard emerging. Prices, let alone contractual structures, are rarely made public, so the following is based on interviews with market players and may not reflect the full diversity of contractual structures. This evolution results from two major trends on the market: a perception of reduced registration risk but of increased delivery risk, and the unexpectedly high price levels in the EU-ETS, at least until late April 2006 (see Convery and Redmond 2007). We discuss the implication of each trend in turn.

First, with the approval of a large number of methodologies for determining whether a project is additional in 2005 and the acceleration of CDM projects registration, the risk of nonregistration—perceived and real—has decreased substantially. Sellers are increasingly confident about selling CERs, and verified emission reductions contracts have all but disappeared. On the other hand, there are indications that performance and delivery risks are perceived with renewed acuity by market players, possibly because some early projects delivered fewer CERs than expected. For example, it is estimated that the CERs that have been certified so far represent 70 percent of the CERs that should have been generated over the same period according to the PDDs (Fenhann 2006). So, CDM projects are still perceived as risky, but for performance rather than registration issues. And registration and performance risks are different. In fact, specialists in the Kyoto Protocol are necessary to manage registration risks, whereas many people and institutions already manage performance risks for specific countries and/or types of projects. As a result, though they are still perceived as risky, CDM projects now attract a wider range of (possibly less carbon-savvy) investors.

To hedge against performance risk, two main routes are being followed. First, complex contractual structures have been built that involve not only carbon but also other attributes, such as, *inter alia*, access to technology, access to debt and/or equity, financial engineering or even agreements to operate projects. Pre-existing relationships between buyers and sellers are of particular importance in such deals. Second, guarantee structures for the provision of CERs, which, as noted above, already existed, have become more common. For example, a creditworthy seller may guarantee that it will replace a fraction of the CERs exchanged in the contract, typically 1/4 to 1/2, in case of nondelivery (e.g., by purchasing compliance units on the market). Such guarantees are reputed to command high premiums (four to six dollars /tCO₂e), and they pave the way for the creation of a secondary market for CERs, as buyers may then immediately sell back the secured credits—which become “quasi EUAs” because of the guarantee—on the EU-ETS market.

The second trend that has affected contractual structure is the price differential between the spot price in the EU-ETS (as high as €30/tCO₂ in June 2005) and the price of forward contracts for CERs (\$3 and \$7.15/tCO₂e for the period January 04 to April 05 (Lecocq and Capoor 2005), which has generated tension on the CDM market. Sellers pushed for better deals, and some even decided to hold onto their assets in the hope of selling at better terms later. In addition, the price differential led to an increase in demand for CERs from firms under the EU-ETS and from speculators who saw opportunities for arbitrage—all that *in*

addition to the increased demand for CERs triggered by the entry into force of the Kyoto Protocol. As a result, the price of CERs has soared, to a range of \$3–24/tCO₂e on the primary market, and of \$21–27/tCO₂e on the secondary market between January 2005 and the first quarter of 2006.

Another consequence of the discrepancy between prices on the EU-ETS and prices on the CDM has been the generalization of contracts in which the price of the CERs is indexed to the price of EUAs. Some indexed contracts include floor prices. Others have ceiling prices beyond which buyer and seller share the upside risk. Initial evidence suggests that projects with indexed prices, especially those without floor prices, suffered strong losses when the price of EUAs fell by more than 50 percent at the end of April 2006. Some small-scale projects, in particular, may no longer obtain sufficient carbon revenues to remain in operation. As a result, indexed pricing may become less common in the future, at least for primary deals.

The overall value of the carbon market was about \$10 billion in 2005, and it is estimated to be \$21.5 billion in the first three quarters of 2006. The EU-ETS accounted for 75 percent of the total market value in 2005 and 87 percent in the first three quarters of 2006. In other words, the total value of project-based transactions was about \$2.8 billion in 2005 and about \$2.4 billion in the first three quarters of 2006, of which 94 percent is CDM (Capoor and Ambrosi 2006a, b).

Development and the CDM

The discussion above should leave no doubt that the CDM will substantially “assist Parties included in Annex I in achieving compliance with their quantified emission limitation and reduction commitments” (KP 12.2). Contracted CDM projects may already supply some 680 MtCO₂e, that is, 17–19 percent of a total expected shortfall of 3.6 to 4 billion tons of CO₂e for Europe, Japan, Canada, and New Zealand over 2008–2012,³ and CDM projects already in the pipeline could already double that amount. But does the CDM meet its other objective to “assist Parties not included in Annex I in achieving sustainable development and in contributing to the ultimate objective of the Convention?”

In terms of *total flows* to developing countries, the CDM certainly plays a small but non-negligible role in providing finance to developing countries. As noted above, the carbon contracts signed in 2005 are worth about \$2.5 billion or 2.5 percent of net ODA for that year. Since carbon buyers are for the most part private companies that are not in the business of investing in developing countries, most of this capital is probably “additional,” in that it would not have gone into developing countries in the absence of the CDM. In addition, by facilitating the financial closure of capital intensive deals, the CDM leverages additional capital into developing countries. The ratio between the amount of carbon finance and the total capital needs of the project is estimated between 1:3 and 1:5 for renewable energy and LFG to energy projects.

³The shortfall is estimated by taking the latest available emission levels for Annex B countries (usually 2004) and applying to these levels the projected annual rate of increase in emissions between 2000 and 2010 as per the latest national communications. The high/low range relate to the “with measure” and the “with additional measures” scenarios in the national communications.

The *nature* of CDM flows is also very important. The fact that payments for CERs usually come on delivery provides higher incentives for the project to perform than traditional ODA. The flip side of the coin is that on-delivery carbon finance provides little relief to project developers often seeking upfront financing to close the financial structuring of their deals. As a result, project developers have had to use carbon contracts as collateral to obtain upfront financing from banks or other financial institutions.

Here, the *quality* of CDM payments helps. Since carbon payments are payable in strong currencies (typically, dollars, euros or yen) and usually originate from buyers with high credit ratings, they are in general more efficient ways to raise capital or contract debt than local purchase agreements. This leverage effect of carbon finance, anticipated by Mathy et al. (2001), has proven effective in some cases. In the Plantar project in Brazil, for example, the PCF is purchasing emission reductions related to the substitution of coal by sustainable charcoal for pig-iron production. To finance the initial investments, Netherlands-based Rabobank granted the pig-iron producer a five-year loan with repayments scheduled against the anticipated carbon revenues. The carbon payments are not made directly to the company, but are paid to an escrow account in a developed country bank, thereby avoiding the two currency risks associated with channeling the money in (to pay for the CERs) and then out of (to service the debt) the country. Pig-iron producers in the state of Minas Gerais typically obtain loans that extend no longer than two years, and the exceptional terms of that particular arrangement are directly linked to the presence of an emission reduction purchase agreement signed by the World Bank.

This project, however, is the exception rather than the rule. In fact, most commercial banks remain unwilling to use carbon contracts as collateral. Niche financial institutions such as mezzanine financiers partially fill the void by granting loans against carbon-purchasing contracts, for example, though quasi-equity or subordinated debt. Some carbon purchasers also offer advance payments of carbon credits, with a significant discount. But if these solutions do contribute to the financial structuring of some projects, the involvement of large financial institutions remains necessary for a large-scale development of the CDM.

As noted above, the *direction* of the flows raises strong distributional concerns about the CDM. China, Brazil, and India account for the majority of the volume traded and the carbon revenues generated by the CDM, while the most capital-constrained countries—most notably those in Sub-Saharan Africa—are left out. Although this outcome is consistent with the distribution of Foreign Direct Investment flows, and may reflect well-known differences in investment climate, it also derives from CDM-related issues. First, Sub-Saharan Africa has a limited supply of large-scale projects, energy or LFG capture, and no HFC-23 or nitrous oxide (N₂O) destruction opportunities. Second, LULUCF activities—the supply of which may be large in Sub-Saharan countries—are *de facto* barred from the market because of their exclusion from the EU-ETS, which not only dries up demand from European firms, but also from non-European buyers who may be afraid that LULUCF credits have a lower resale value on the secondary market.

Of course, financial transfers *per se* do not constitute sustainable development. What matters is how these resources are used. Here, CDM projects may contribute to sustainable development in two ways: the project activity may contribute directly to sustainable development, and/or the revenues that the project generate may be recycled into activities conducive to sustainable development. Most of the literature focuses on the first of these

two connections (i.e., the direct contribution of CDM projects). In fact, several methods have been proposed to assess the sustainable development “score” of CDM projects (e.g., Thorne and La Rovere 1999; Sutter 2003; The Gold Standard 2006). Overall, there seems to be broad consensus to consider that renewable energy and energy efficiency projects tend to contribute to sustainable development. On the other hand, HFC-23 or N₂O destruction are generally considered to have very little sustainable development impact per se. In the middle is a large gray zone of controversial classes of projects, such as LFG capture, hydro or LULUCF, that are hotly contested.

But the direct contribution of a project to sustainable development is not the end of the story. The rents generated by the project may also be used in a manner conducive to sustainable development. For example, the 65 percent tax on the emission reductions generated by HFC-23 projects levied by China will go into a special fund to finance climate change-related activities. Similarly, the additional revenues that municipalities draw from LFG capture may be used to improve municipal waste collection services. The recycling of CDM rents, however, remains virtually undocumented at this time. This makes the overall contribution of CDM projects to sustainable development very difficult to assess.

Sustainable development can also be approached from a *procedural* point of view. For example, does the Letter of Approval by DNA constitute a sufficient test of sustainability? Or does the CDM provide for fair consultation of stakeholders? There has been a substantial amount of debate on these issues, and the responses in the literature seem overall to be negative. For example, Beg et al. (2002) question DNA capacities to assess whether a project indeed meets the country’s own sustainable development criteria. Burian (2006) proposes improvements to the stakeholders’ consultation process so that it is more inclusive and provides for more back-and-forth discussions. The aforementioned NGO-backed standards for CDM projects all propose some degree of procedural improvements as well.

Finally, the impact of CDM projects must also be discussed dynamically. Because a project has to be additional to what would have happened otherwise, the CDM may provide a strong disincentive for countries to adopt environmental policies that would make some activities ineligible for the CDM. However, the presence of a regulation has not always been interpreted as making the project nonadditional, as long as it can be demonstrated that the regulation is not followed in practice (e.g., by looking at comparable projects around the country). In addition, two decisions by the EB make it clear that the existence of regulatory frameworks that encourage the implementation of clean energy projects, such as renewable energy targets or feed-in tariffs, may not be taken into account in developing a baseline scenario, provided they have been implemented after the adoption of the Marrakesh Accords in November 2001. But the risk still exists that a country will not pass “good policies” in order not to lose the opportunity to attract CDM resources. This calls for periodic reassessments of baselines at the country level.

Unresolved Issues and Future Prospects

The CDM is the result of a compromise between Annex 1 Parties eager to get access to lower-cost emission reductions in developing countries and developing countries eager to get additional financing for development. Nearly a decade later, the bargain has been met,

to a degree. On the one hand, Annex B countries have gained access to emission reductions at lower prices than domestic abatement costs, although the market price of CDM assets is substantially higher than expected. On the other hand, major emitters among non-Annex B countries will receive substantial carbon revenues. Although a handful of chemical producers (and the Chinese Government with its tax on HFC-23 project) extract most of the rent, it can be argued that the CDM has contributed to keeping developing countries, and in particular the strategically important large emitters among them, involved in the global carbon market. The CDM has also contributed to increasing awareness about mitigation in developing countries, and given a large number of stakeholders in the developing world a sense of involvement in the Kyoto Protocol. This outcome was far from obvious after the Kyoto negotiations, during which the G77 and China were mostly sidelined by Europe and the United States. And from a development perspective, the CDM has proven, albeit on a small-scale, that it can channel substantial private capital into clean energy projects, with strong leverage effect.

From the perspective of the political economy of the Kyoto Protocol, the CDM may also provide the EU and Japan the necessary flexibility to actually meet their targets without relying too heavily on politically unpalatable purchases of hot air—that is of ‘excess’ AAUs from countries, such as Russia or Ukraine, which are likely to emit much less than their Kyoto targets between 2008 and 2012. The issue of greening hot air, however, remains critical given that the CDM is not likely to supply more than 20–25 percent of the total expected demand for CERs (World Bank 2004). The ultimate success of the Kyoto Protocol still remains very much in the balance.

Looking ahead, it is becoming increasingly clear that meeting the climate challenge will require substantial cuts in emissions in developed and developing countries alike by the middle of the century. Of critical importance is avoiding the lock-in of long-lived capital stocks—yet to be built in China and India—into carbon intensive paths. Doing so will require the implementation, on a very large scale, of a combination of policy changes (e.g., power pricing reforms), technology transfers, investment, and possibly complementary international agreements to, for example, guarantee safe access to energy resources (Heller and Shukla 2003; Hourcade and Shukla 2006; Newcombe 2005; Victor 2006). A comprehensive approach will also be required to address deforestation—which accounts for twice as much emissions as transport (Chomitz 2007).

The CDM is currently ill-equipped to fulfill this task. The additionality concept, in particular, may be impossible to operate for countrywide policy reforms. In addition, bringing about systemic change requires concentrated action in a single sector or country, whereas at the heart of the CDM is the logic of targeting the lowest cost opportunities regardless of where they appear.

Does this mean that there is no scope for the CDM in the future? We do not believe so. If large-scale agreements on clean energy are a priority in large developing countries (China, India, Brazil, Mexico, possibly South Africa, Korea and maybe a few others), the vast majority of developing countries have emissions that are simply too low to justify a separate treaty or instrument. Yet these countries often have non-negligible emissions, are usually capital-constrained, and could thus well remain involved in an (improved) CDM in the near future.

Three issues, however, need to be resolved. First, additionality remains a weak link. On the one hand, there are indications of strategic behavior by chemical producers to increase revenues from CERs (Wara 2006), an extremely disturbing finding given the predominance of these projects in the CDM portfolio. On the other hand, the additionality test remains an obstacle to the development of grid-connected clean energy projects (such as fuel switching or renewable energy) over individual gas destruction projects.

A related issue is that project-by-project analysis does not allow for the consideration of policies and programs under the CDM, and may even lead to inconsistent results as individual baselines over the same grid have little reason *a priori* to be mutually consistent. The development of sectoral approaches to the CDM, such as sectoral benchmarks, with less-constraining requirements for the additionality of clean energy projects is a possible way forward (e.g., Schmidt et al. 2006).

Third, the dynamic effects of the CDM on regional or national policies have to be addressed. Since there is no way to determine what policy is additional and what is “business-as-usual,” negotiations over CDM projects should directly internalize the policy component by treating it as part of the negotiation package.

Our final point is that maintaining the momentum of the CDM until 2012 may not be easy. The growth of the CDM market has been remarkable over the past few years, but as we get closer to 2008 or 2009, the window of opportunity to register projects will close because the time necessary to finalize the transaction and build the project will be too long to generate any meaningful emission reductions before the end of 2012. Yet it is not clear that the post-2012 climate policies will be agreed upon by that time (in fact, negotiations on post-Kyoto climate policies will start only in 2008), and without a clear price signal for post-2012 credits, buyers are much less likely to invest in CDM projects. The risk is thus that the CDM will lose momentum in that interim period, even though the voluntary market and several regional initiatives in the United States and Australia may to some degree support the CDM market.

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Reflections on the Literature

V. Kerry Smith*

Introduction

Starting a new venture is never easy, and this inaugural column has certainly been no exception. The task seemed especially difficult when I realized that the editors were hoping my column would present something you might actually be interested in reading. When they recruited me to prepare this column on the current literature, the editors suggested that I pull ideas from published articles, working articles, policy issues, new data sources, or anything else likely to attract some interest. I hope that in the future some of the ideas will come from you.

This time my selection is dominated by the environmental side of resource and environmental economics. I have prepared short discussions for each of three research areas. Each one offers conjectures about existing research along with suggestions intended to elicit your reactions. No doubt, some of my comments may seem “half-baked,” but the point is to get you thinking. If they do that, then I have accomplished my goal. As a result, I expect to hear some responses and will try to reflect them in future columns.

The areas discussed in this first “Reflections” are behavioral economics and policy, the Tiebout model—old insights and new implications, and the limits to partial equilibrium benefit-cost analysis. In each case, the discussion considers one or more articles that present a framework for analysis, a set of policy recommendations, or new empirical findings. Each topic was selected because it is interesting to me. Hopefully, you will agree.

There are several reasons for selecting the first topic, which focuses on behavioral economics. This research has offered a rich set of challenges to conventional theory and with them promising amendments to how we describe consumer choice. Fudenberg’s (2006) recent review article on “Advances in Behavioral Economics” has carefully outlined the issues the “field” of behavioral economics must confront before being regarded as part of the corpus of mainstream economic analysis. My goal here is more narrow. It concerns whether the insights from behavioral economics are ready for the prime time of policy evaluation—a big jump from entertainment news programs.

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With the recent changes in the default option for 401K plans motivated in part by the strongly worded recommendations of behavioral economists, it seems reasonable to ask, what's next? Advocates of the behavioral perspective contend that their insights do call for quite different perspectives on how economic analysis is used to inform policy. Consumer sovereignty is questioned. Guided choice seems preferred. Do we have widespread acceptance of this position? If so, how should policy evaluations be conducted? Bernheim and Rangel (BR) (2007) argue a change is warranted and behavioral economists have the answers as to how public policies need to be evaluated. In my view, the state of the art is very far from ready. So I begin by considering whether people's mistakes warrant a new paradigm for policy analysis.

The discussion of Tiebout in the third section is tame by comparison. Here, consumer sovereignty is not only trusted—it rules! Different people are expected to notice areas where there is a potential to reduce a cost or to realize some gain, regardless of whether these effects arise in markets. With these opportunities, as well as individual differences in ability and willingness to adjust, we can expect that these responses will influence the performance of many types of policies.

If you do not believe adjustment of this type is as important as a price adjustment, consider some examples. Did you vote in our recent midterm election as the polls opened, in the middle of the day, or did you request an absentee ballot? How do the undergrads at your institution get their football and basketball tickets? At Duke, they camp out in front of the sports complex for home basketball tickets days before key home games. With fixed (and relatively small) arena capacity, the university could easily allocate the tickets by a lottery. Coach K. prefers a system that relies on tent city queuing—why? What does this system do to the composition of fans at home games? Should we consider these types of adjustments outside markets? In some cases do they influence other market and non-market outcomes? My answer is yes! What is especially interesting about this new research is that it is assembling a set of models that allows these types of analyses to take place.

In the context of housing or other spatially delineated consumption goods, a person can “purchase” site-specific amenities by selecting a place to consume the services of specific types of private goods. For a house, it is a neighborhood. Of course, the logic assumes that a full set of jobs will be freely available with about equal proximity to each location. We know this is not the case. People must balance competing needs that each have implications for spatial choices. If they have enough money, some people may simply choose to purchase several spatially differentiated goods. Section three ends with a brief discussion of second homes and what they imply for nonmarket valuation using the Tiebout and hedonic models.¹

The fourth section considers the supersizing of benefit-cost analyses. In twenty-five years, benefit-cost analysis has gone from a backwater to a business. Today we take for granted that partial equilibrium approximations can be used equally well for evaluating small projects and large ones. This section asks, but does not answer the question—how big is too big?

Finally, the last section closes with some discussion of the philosophy for this column.

¹ In this case I am referring to a hedonic property value model, where we assume that the process of matching buyers and sellers for homes that are differentiated by their structural and locational characteristics leads to an equilibrium with no incentives for anyone to change their decisions. That equilibrium implies the homes' prices will vary with these characteristics and we routinely approximate it with a function.

Modeling Mistakes

Sometimes it seems the current literature abounds in efforts to find people's mistakes. Individuals who are confronted with what appear to the analyst to be obvious choices do not take them. Instead, they make mistakes. Behavioral economics has suggested these patterns warrant a different approach to policy recommendations—one that selectively accepts consumers' choices. This research is now calling for a full-scale reevaluation of conventional microeconomics and its role in providing information to the policy process. Consider any graduate microtext—it begins with the assumption that people know what they are doing. In fact, as a rule, we are prepared to assume much more than that. For example, Mas-Colell, Whinston, and Green (1995) on the first page of the first chapter of their graduate microtheory book observe that consumer sovereignty is at the core of what we assume in developing economic models for behavior. Individual choice behavior can be represented with preference relationships that impose rationality axioms, or it can be derived from the weak axiom of revealed preference. Regardless of the logic used, our economic analyses rely on judging outcomes based on the trade-offs revealed through people's choices. Behavioral economics is arguing this premise should change.

There is a wide spectrum of literature included under the mantle of behavioral economics. I will focus on the recent work that suggests enough has been learned to modify the way economics is used in policy analysis. For example, some authors such as Bernheim and Rangel (2007), "argue it is sometimes possible to replace revealed preference with other compelling normative principles . . . if one knows enough about the nature of decision making malfunctions, it may be possible to recover tastes by relying on a *selective* application of the revealed preference principle" (p. 9). Going a bit further, Thaler and Sunstein (2003) "defend libertarian paternalism, an approach that preserves freedom of choice but that authorizes both private and public institutions to steer people in directions that will promote their welfare. . . . The goal should be to avoid random, arbitrary, or harmful effects and to produce a situation that is likely to promote people's welfare, suitably defined" (p. 179).

Avoiding random, arbitrary, or harmful effects and taking actions that promote people's welfare—this proposal seems like a clear-cut winner! Unfortunately we cannot resolve the problems that these authors suggest arise when people do not make what they (the authors) think is the appropriate choice. There is one small detail. Who decides what is a *suitably defined* welfare improvement—elected officials, policy makers, experts? If experts are given this authority, then do they need to understand people's decision-making quirks, the scientific and technical details of each and every activity that policy might involve, or both? This seems like a tall order. We may need more than one type of expert. This prospect then leads to another problem—how will these experts communicate with each other? Communication, after all, is one of the sources of the problems laypersons and experts face in making choices about things outside their domain of experience. The information available to them is precluding "appropriate" choices. Experts will be able to evaluate their own fields but they would have to know each other's fields to form trade-offs between complex objects of choice. Fortunately, in the view of those behavioral economists who are arguing that they are ready to offer prescriptions for policy, there are no problems, really. We can engage a trained facilitator, a communication expert who can ensure that each group's position is appreciated, before we arrive at a public choice that promotes people's

welfare. Only under these conditions could we be sure that people's welfare was "suitably defined."

I suspect this process is beginning to sound complicated (and that, of course, was my intention). Policy makers would have to select each set of experts. But wait, the policy makers may need help. After all, they have their own limitations. At this stage, I'm sure the reader sees my point that the process is cascading through one set of experts to the next, and as a result, is never-ending.

On the opposite side of the spectrum from these advocates for using what has been learned to transform economists' recommendations to policy makers, we have Glaeser's (2006) recent reactions to Thaler and Sunstein. He states, "if humans make mistakes in market transactions, then they will make at least as many in electing representatives, and those representatives will likely make mistakes when policymaking" (p. 32). We could extend his argument to conclude that these mistakes could include choosing the wrong experts to listen to.

Much of environmental economics is about the design and evaluation of policy. As a result, these debates are especially relevant to environmental economists. The acceptance of consumer preferences is rarely questioned. Analysis of questions dealing with environmental risk and ecological systems might be the only exceptions. And even in these cases, most environmental economists would argue that the problems they pose are largely associated with delivering understandable information to the lay public about a specific risk or about the multiplicity of services generated by ecosystems. The conventional literature does not question consumers' capacity to deal with the information when they understand it. Moreover, it does not ask whether their choices are consistent with suitable definitions for their well-being.

It seems clear that behavioral economics undermines any special rationale we might propose for using information from a benefit-cost analysis to inform policy. If measures of people's trade-offs between private resources and the services that are expected to be provided by adopting a public rule are questionable, it hardly makes sense to compare an aggregate of these measures of incremental willingness to pay to the costs imposed by the rule. A hard-won "place" at the policy table is up for grabs if we accept their proposals for changing our approach to policy evaluation. Many experts can offer their personal definitions for what is "suitable" for consumers. Moreover, behavioral economics calls for what Thaler and Sunstein have labeled "soft paternalism." This position would argue that our objective should be to persuade the lay public to accept what the policymaker thinks is best.

This is not information and choice. It is advertising. As Glaeser's (2006) article notes: persuasion is central to soft paternalism. People should not want policymakers to become adept at persuading us to invest in infrastructure supporting that persuasion. Where will it end? Abuse seems likely.

Thus, for both conceptual and policy reasons, a closer look at behavioral economics within environmental economics seems warranted.² BR's chapter in a new book, *Economic Institutions and Behavioral Economics*, is an excellent starting point. The Bernheim-Rangel article identifies two schools of thought for the behavioral conceptual model—what I have labeled here as the framework for "modeling mistakes." The first, they suggest, insists on

²Of course, I realize much of this discussion has seen active participation from environmental economists. My point here is that the newest overviews seem to have overlooked this work, and it is time environmental economists were engaged in these discussions.

strict adherence to revealed preference theory. It proposes to amend preference space so that analysts can explain “anomalies” and thus “robustly rationalize choice.” The second is willing to relax or even dismiss the revealed preference logic. When people make systematic mistakes in identifiable circumstances, this perspective suggests that it is appropriate to selectively ignore the revealed preference outcomes.

Thus, this second school of behavioral economics modifies revealed preference, accepting it when it is coherent—not too many mistakes—and replacing it when it is not. The skeptic should ask who decides what choices are mistakes and when enough are made so that the evidence compels a conclusion that preferences in these dimensions are incoherent. What is the response to “faulty” preferences, training or practice? Do we simply offer more or better information? Do we impose a “behavior tax?” Can we label such a tax as the shadow price equivalent of a “time out” for bad choice behavior? We cannot deny that laboratory and field experiments suggest people can make silly mistakes. How should we respond? This is the issue we should consider.

The BR overview highlights three areas where behavioral economic analyses have been especially influential: (a) understanding savings behavior, (b) modeling of consumption of addictive substances, and (c) contributing to the creation of public goods. Their comments on the first and last areas of research are especially relevant for environmental economists.³ Most experience with actual choices as well as laboratory experiments indicates that choices (and the trade-offs they imply) are different when one of the objects of interest is consumed in the present and is being compared to a resource available in a future time versus when both are in the future and are separated by the same amount of time as in the present to future case.

Hyperbolic discounting tries to explain this discrepancy. It assumes that the trade-offs in situations where both sets of consumption are in the future will adhere to a conventional form of discounting. By contrast, it maintains that when one of the time periods is the present, the trade-offs do not adhere to conventional discounting. Certainly, this line of work has attracted a lot of attention among environmental and resource economists. It has been shown to be especially important for long-term policy actions, such as climate change. More research is definitely warranted. Time can be considered an attribute of goods and services, so we need not dismiss rational choice in order to acknowledge that people may evaluate consumption—especially for nonmarket resources—in ways that are different from financial markets.

What is less well recognized and developed is a model that suggests cue-triggered mistakes, i.e. people are sophisticated and rational some of the time and other times they are “faulty.” The implication of this formulation is that precommitment and managing the cues or “triggers” to bad behavior are strategies for improving behavior. To my knowledge, these types of issues have not been considered in models for “bad” environmental behaviors.

BR’s discussion of public goods is also informative for another reason. Little of the extensive work on closely related concepts from public and environmental economics has been recognized by behavioral economists.⁴ When the source for warm glow, nonuse

³See Karp (2005) for a good example of this line of research.

⁴List’s research is a notable exception. List (2001) is widely cited in this literature.

values, or some of the definitions for altruism is introduced into individual preferences, the resulting models can imply links in how private allocation choices increase several individuals' levels of well-being. As a result I would expect that more active discussion and debate between environmental and behavioral economists, about what best characterizes the linkages between some types of private and public goods in preferences, will improve both sets of research.

Where will the modeling of mistakes take us? Is it likely to induce theorists to revamp our model of rational choice from the ground up? I doubt it, but that is just one person's opinion. It does challenge a central strategy of research in the neoclassical framework. That is, we are taught to develop models that explain as much as we can with minimal assumptions about the black box that comprises personal choice. As Lucas (1986) suggested, in response to criticism of how superficial the economic model of human behavior is, in economic theory, "it is exactly this superficiality that gives economics much of the power that it has: its ability to predict human behavior without knowing very much about the makeups and the lives of the people whose behavior we are trying to understand" (p. 425). It seems to me that learning more about the "black box" would be a good thing. But do we all need personal laboratories with MRIs and support technicians? I doubt it.

Tiebout at Fifty⁵—Two New Directions

Fifty years ago Tiebout (1956) suggested that the demand for local public goods was revealed through household locational decisions. Differences in community conditions offer an opportunity for people to select nonmarket goods—the public goods that vary across communities. Environmental amenities can be a large part of these differences. Households reveal their relative preferences for these services by changing communities. Oates (1969) raised the profile of this model for local public finance. Two extensions to Tiebout's logic can be found in recent research. For one of these, a lot of research has accumulated in a relatively short time. For the other, the landscape seems to be less developed. Both are worthy of more attention.

Sorting Models and Nonmarket Equilibria

Structural econometric models of household sorting can be seen as reflections of the Tiebout model's basic insights. They are of interest here because these models also offer opportunities to examine household and firms' responses to feedbacks that arise outside markets.

To illustrate what I mean by these potential feedbacks, consider the case of congestion. We all know that people learn when highways will be congested, and adapt. However, I would not have expected this point to arise in a discussion about the strengths and weaknesses of public warning systems for hurricanes. During the course of focus groups in Houston with the evacuees from Hurricane Rita, Carol Mansfield and I heard that some of the participants had used Mapquest to find alternative routes out of the city in order to avoid the recommended evacuation routes, because they anticipated that the public evacuation

⁵The new Lincoln Institute book to honor Wally Oates, *The Tiebout Model at Fifty*, edited by Fischel (2006) prompted this title. Fischel's introductory essay has an interesting history of Tiebout's professional life.

routes would experience high levels of congestion. The experience of those who innovated and did not “follow the rules” was quite different from those who did. People take these types of actions every day in commuting, or in planning their trips to recreation sites, or in a host of other actions. In these cases, there is not an explicit price signal that triggers adjustment. It is something else. Particular roads and commuting times are better than others. The point of the evacuation story is that responses to a policy may lead to equilibria that reflect adjustment we might not anticipate if our analysis focused exclusively on shadow values that are easily converted to monetary equivalents, such as the opportunity cost of time. A variety of signals of the consequences of social interactions may create responses that are different to what might be expected from a more conventional interpretation of the policy.

This point is not limited to congestion and traveling by car. Some time ago Becker (1991) suggested that crowding may be desirable from the perspective of a restaurant. Owners may wish to limit capacity knowing that customers use waiting times and full capacity as a signal of a restaurant’s quality. What is not discussed in this argument is the adjustment that can take place once quality is recognized. Some people will adapt, recognizing that a popular restaurant may not be busy every day. Summer patterns of consumption are different in winter. Daylight savings time causes adaptation. Thus, nonprice adjustments arise in many different ways and we are only beginning to recognize the important role they can play with environmental resources.

In the mid-1970s, Dorfman (1974) defined common property resources as conduits for externalities—they created ways in which interaction among different people would lead to positive and negative interaction. His point seems more general. Not only do the resources convey externalities but they also could signal the consequences of different patterns of use that would lead to private adjustment. This is not news; the point abounds in the literature on external effects and is an important reason why optimal public pricing (or taxing) models make strategic assumptions that limit how the choice of a private good relates to the level of a public good or bad that may not be under a person’s control. What I think is new is the recognition that we can begin to use the structural models describing sorting behaviors to compute the equilibrium responses to these nonmarket effects. Whether it is fishing and catch rates (see Newbold and Massey [2006]) or the number of public vaccinations and private protection (Geoffard and Philipson [1997]), the logic of a feedback mechanism is comparable. Thus, these new applications of the Tiebout logic are a serious effort to build the capacity to include heterogeneity, at the individual level, into models capable of simulating the outcomes of new policies when feedback adjustments can be assumed to be present.

The first approach to sorting models by Epplé and Sieg (ES) (1999), develops a structural econometric model of households’ choices of communities, based on the logic outlined in earlier contributions by Epplé, Filimon, and Romer (1984) and Epplé and Romer (1991). The economy consists of a continuum of households. They select from among a finite number of communities. Communities are defined by the public goods available within each of them. As part of the details of making the model operational, communities can be given spatial boundaries, such as school districts or the GIS boundaries for census block groups. The criteria used to define each of the spatial units depends on the goods or services the analyst assumes a household obtains by selecting a particular community.

More generally in other contexts, these alternatives can be as diverse as recreation sites or evacuation routes. In each case, an individual is trying to choose one type of good or service from among a finite set of alternatives.

Simple versions of the model used to describe households' choices of communities often assume a finite supply of housing. Households differ in their income and preferences for public goods. This preference difference is represented with a single parameter that is assumed to be a source of unobserved heterogeneity. When households can move costlessly, an equilibrium implies that no household wants to move. The ES framework uses a specific algebraic form for preferences. It must satisfy conventional properties, such as diminishing marginal rates of substitution, nonsatiation, and monotonicity, along with what is described as the single crossing property. This condition assures that the relationship between a change in a local public good and the price of constant quality housing will undergo a predictable change with increases or decreases in household income.

Under these conditions, the attributes of the equilibrium, together with a specific structure assumed for preferences for public goods, assure that the model has predictions for housing prices across communities and for how people will sort themselves among these places. Epple and Sieg have shown how these predictions can be used to estimate the parameters of underlying preferences. Sieg et al. (2004) have used this model to evaluate the effects of exogenous air quality changes on partial equilibrium and general equilibrium benefit measures that have the potential for feedback effects.

A second approach, developed by Bayer, McMillan, and Rueben (2005), is also a sorting model, but it relaxes important assumptions about preferences for public goods. In the ES model, people must evaluate the contributions that specific public goods make to the overall index of public goods provided by a community in exactly the same way. This is referred to as vertically differentiated preferences. In contrast, Bayer et al. allow different people to have different ways of evaluating public goods. This is referred to as horizontal differentiation.

Both models allow predictions of the adjustments that are made in response to exogenous changes. And here is where the potential for a new line of research arises. Suppose a change in something within one or more communities causes households to adjust, but one part of why they select a community is due to things that happen because others are also there. Under these conditions, depending on what we assume, the model allows the analyst to include a separate, nonmarket signal that is a feedback effect stemming from these nonmarket interactions between households. This influence affects the equilibrium in prices for houses and the nonmarket outcome. Epple, Romer, and Sieg (2001) suggested a model for local decisions about the levels of community-specific public goods—the median voter model. Timmins and Murdock (2007) proposed the same type of logic for modeling the choice of recreation sites. They postulate that congestion affects the attractiveness of a recreation site. For example, a policy that stocks a fishing site may cause more anglers to come, thereby increasing congestion. However, if we assume that most recreationists are aware of the congestion, then some might not come. The pressure on the fish stock is not as great, depending on what we assume about the stocking and natural population growth. Ultimately, the outcome (in a Nash equilibrium) will depend on what we assume about preferences for congestion, as well as the population dynamics of the fishery! The

equilibrium should reflect this adjustment and so should the benefits from the proposed fish stocking activity.⁶

Are there additional avenues for research on feedback that influence behavior? Yes, many. Walsh (2007) used similar logic in evaluating policies to protect open space. In his analysis, open space is composed of public and private undeveloped land. In this situation, one finds that the purchase of private land for permanent protection may increase demands to be in communities with such new areas. However, this increased desirability can actually cause the private undeveloped land in a community to be developed and overall open space to decline!

Recognition of feedback effects in market outcomes has been a part of our description of the equilibrium dynamics of the price adjustments in market processes for decades. This new work reminds us that this logic also applies to nonmarket feedback. It allows the potential for differences in people's responses to nonmarket services they value to influence market outcomes. Further, this logic acknowledges that these influences may, in turn, determine the nonmarket service we wind up getting. Thus, there seems to be clear potential for a closer integration of market and nonmarket allocations.

Policies that recognize heterogeneity in preferences (or in the ability to respond to incentives) have been a part of modern public and environmental economics. However, it has been difficult to evaluate, *ex ante*, their implications, especially when they are intended to influence a nonmarket outcome. Whether it is regulating snowmobiles in Yellowstone, franchising commercial carriers for raft trips in the Grand Canyon, vaccination policies for children or evacuation plans for hurricanes, the differences in how people respond to each policy will influence the performance we attribute to these interventions. These Tiebout-esque models offer us the capacity to begin to consider how heterogeneity affects nonmarket equilibrium adjustments.

Tiebout and Second Homes

In June 2006, the Lincoln Institute for Urban Development sponsored a conference with a wide range of very interesting articles. One of them, by Belsky, Zhu, and McCue (2006), considered the implications of individual household ownership of multiple homes for measuring the income elasticity of housing demand. This is an interesting question. However, their discussion raises a separate but equally interesting question: how do second homes fit in the Tiebout logic?

Historically, second homes have grown in importance. The 1965 Census (in the Housing Vacancy Survey) reported 2.6 million homes usually classified as second homes. This was about 4 percent of the total housing units. By 2005, the total number of housing units had nearly doubled and second homes were about 5.5 percent of the total. Based on 1990 data reported in Zhu et al. (2001), half of the second homes were found in just 150 of the 3,000 counties in the United States. A quick check of the current census did not offer detailed data at the county level, but it does provide a clear indication that for some states, seasonal,

⁶We have a long history of recognizing the potential to use heterogeneity in price responsiveness in the design of policy. In the context of nonlinear budget constraints, see the early article by Blinder and Rosen (1985) as an example. The distinction here is in the heterogeneity in responding to nonprice signals.

Table 1 Ten states with the highest percentage of seasonal, recreational, or occasional-use homes, 2000

| Area | Total housing units | Housing for seasonal or occasional use | Percentage |
|---------------|---------------------|--|------------|
| United States | 115,904,641 | 3,578,718 | 3.1 |
| Maine | 651,901 | 101,470 | 15.6 |
| Vermont | 294,382 | 43,060 | 14.6 |
| New Hampshire | 547,024 | 56,413 | 10.3 |
| Alaska | 260,978 | 21,474 | 8.2 |
| Delaware | 343,072 | 25,977 | 7.6 |
| Florida | 7,302,947 | 482,944 | 6.6 |
| Arizona | 2,189,189 | 141,965 | 6.5 |
| Wisconsin | 2,321,144 | 142,313 | 6.1 |
| Montana | 412,633 | 24,213 | 5.9 |
| Hawaii | 460,542 | 25,582 | 5.6 |

Source: U.S. Census Bureau, Census 2000 Summary File 1. (For information on confidentiality protection, nonsampling error, and definitions, see www.census.gov/prod/cen2000/sf1.pdf.)

recreational, or occasional-use homes account for over 10 percent of the total housing units (Table 1).

Are second homes being used as a way to “acquire” amenities that households cannot obtain at their primary residence? If so, who is making these choices and what do these decisions imply for the Tiebout logic that assumes choosing a community is the only way a household can acquire the public goods involved?

Recognition of the connections between markets in adjusting to site-specific amenities is not new to environmental economics. Rosen (1979) and Roback (1982) demonstrated how hedonic property and wage markets can be related.⁷ McConnell (1990) and Parsons (1991) pointed out that similar connections can influence how we interpret the amenity values from hedonic and travel cost models.

The growth in second homes suggests that there may be other, largely unexploited opportunities to learn about specific groups by considering these alternative markets. Moreover, it also implies we may well derive an incomplete picture of the intensity of the demand for some amenities by only considering decisions associated with a household's primary residence. This seems to be an under-researched area, at least from the perspective of its implications for environmental economics.

Carlinger (2002) reports some more specific information about second homes that would seem to support my call for further research. For example, second homes are primarily held for recreational purposes. Between 1999 and 2001, the holding of second homes for recreational purposes increased while other explanations such as investment, unable to sell, or inherited properties declined. Data from the American Housing Survey in 1995 (see Zhu, McArdle, and Masmichs [2001]) indicate that households headed by individuals aged 55–64

⁷See Blomquist et al. (1988) for the most detailed reduced form application of this logic.

were more likely to own second homes than those with either younger or older heads. However, this pattern may simply be a result of the income and wealth levels for this group. There is some evidence that the prospect of owning a second home is spreading to younger households. The fraction of the 45–54 age group owning second homes closely follows the older group (i.e., 3.99 percent for the 55–64 age group versus 2.82 percent for the 45–54 age group).

Given the concentration of these second homes in a few states, as suggested by Table 1, it would be interesting to pursue the geographic extent of the market for second homes. Another potential question concerns the effect that owning a second home has on how we should measure both the amounts of amenities a household consumes and their willingness to pay for improvements in different types of amenities.⁸

Supersizing Benefit-cost Analysis

Ten years ago, a team of ecologists (and economists) estimated the annual willingness to pay for some seventeen ecosystem services from sixteen biomes around the globe, and concluded that the earth's ecosystems had an annual economic worth of \$33 trillion dollars (Costanza et al. 1997). This estimate was about 1.8 times the world GDP and, in the words of one critic, was “a lower bound on infinity.”⁹ Its media appeal did not diminish the economists' criticisms. However, critical comments did not receive equal space. Recently, the University of Vermont announced that the Gordon and Betty Moore Foundation will now assure that the Costanza team can give “a sophisticated portrait of the ecosystem dynamics and value for any spot on earth.” The press release announcing the award notes that “conventional economics has relied on the rather *clunky* notion of ‘willingness to pay’ to determine how much a product is worth” (italics added). Costanza is quoted as noting that they will be “looking for effects of ecosystems on [*sic*] human welfare, whether people perceive them or not.” No doubt their new results will attract comparable media attention and, based on Costanza's comment, it looks like we might have another candidate for who would be willing to specify what comprises a suitably defined welfare improvement.¹⁰

At about the same time as the release of the report on the earth's value, the first of EPA's mandated reports (U.S. EPA 1997) on the net gains from the Clean Air Act, labeled the Retrospective Study, became available. It reported new benefits from the ambient air quality changes attributed to the Clean Air Act between 1970 and 1990 of \$21.7 trillion (in 1990) dollars. This evaluation attracted less media attention than the Costanza team's *Nature* article but it was also questioned by environmental economists. However, the EPA report should be interpreted differently from the Costanza analysis. The EPA estimate is the value in 1990 for all of the improvements in air quality between 1970 and 1990. While the

⁸Carlinger (2002) reports, based on the 1999 American Housing Survey, that nearly half are within 150 miles of the primary residence, 37.4 percent are 150 miles or more away, and 13.9 percent are not reported.

⁹I am not sure of the exact source of this most appropriate characterization. Several different economists have told me that David Pearce, David Simpson, and Michael Toman were independently responsible for offering this commentary in reacting to the article. Whoever turns out to be the first source of this characterization, we can all agree “it hits the nail on the head.”

¹⁰This could be an example of ecological libertarian paternalism. We will need to wait for the results to be sure.

net benefit estimate is nearly three times the 1990 GDP of \$7.1 trillion (in 1990) dollars, it should be thought of as the value of the environmental assets produced over the period.¹¹ Thus, a more appropriate comparison would consider how the annualized value of the asset, some \$2.17 trillion (in 1990) dollars, using the 1990 prime rate, aligns with the GDP. This is about 30 percent of the GDP or just under \$20,000 dollars per household in 1990.¹²

Now, in a 2006 article in the *Journal of Political Economy*, two University of Chicago economists, Murphy and Topel, are suggesting what they characterize as enormous economic values for the improvements in health and longevity over a thirty-year span from 1970 to 2000. They observe, “From 1970 to 2000, gains in life expectancy added about \$3.2 trillion *per year* to national wealth, with half of these gains due to progress against heart disease” (p. 872, authors’ emphasis).

These estimates are in 2004 dollars, so the comparable annual values for ecosystems and the Clean Air Act are \$47.7 and \$3.1 trillion respectively. No matter what aggregate income base we use, these are big values. Time is overdue for us to ask: do any of these analyses make sense? My guess is that they do not!

Murphy and Topel offer a creative and interesting extension to the logic used at an individual level to estimate the economic value of improvements in health and reductions to mortality risks, but that is not the issue. The estimates are applied as if the individual willingness to pay measures can be “scaled up” for the economy as a whole—similar in spirit to the other two studies I cited. It is important to be clear about my concern. Murphy and Topel do not apply the value of a statistical life (or equivalently, the *ex ante* trade-off between wages and changes in survival probabilities) as if they were constant over the life cycle. Instead, they calibrate individual preferences and the shape of the time profile of marginal values for life years as a person ages. My concern arises because the parameters influencing each individual’s choices, and with them these incremental values, change with the policy being evaluated. We cannot assume the market and the nonmarket constraints on the optimal time profile will be invariant. If all people experience the same change, then the circumstances influencing choices are endogenous.

This point brings us back to the interaction of market and nonmarket influences in determining general equilibrium outcomes. For some situations, the effects of ignoring these interconnections are unlikely to be important. But for changes of this scale, we cannot assume a partial equilibrium perspective is “good enough” for a policy evaluation. When economists take on the “big” questions like the merits of large-scale public and private

¹¹The EPA report describes the procedure used to compute the benefits as follows:

The total monetized economic benefit attributable to the Clean Air Act is derived by applying the unit values (or ranges of values) to the stream of monetized physical effects estimated for the 1970–1990 period . . . The economic benefit estimation model then generated a range of economic values for the differences in physical outcomes under the control and no-control scenarios for the target years of the benefits analysis: 1975, 1980, 1985, and 1990. Linear interpolation between these target years is used to estimate benefits in intervening years. These yearly results are then adjusted to their equivalent value in the year 1990 and summed-up to yield a range and mean estimate for the total monetized benefits of the Clean Air Act from 1970 to 1990 (p. ES-8).

It appears they were capitalized in 1990 using a 5-percent discount rate.

¹²This is based on an estimated 108.7 million households in 1990.

investments in health care or the environment, then these analyses must acknowledge that the changes involved are not marginal programs. They can transform resource allocations for the macro economy and with such changes produce new sets of relative prices and incomes. Thus, the Costanza-proposed research program, the EPA Retrospective Analysis, and Murphy and Topel's evaluation, together present a common challenge: can we say something about large-scale, economy-wide changes? For this reviewer, the jury is still out.

This Column as a Conversation

It is not clear how one should close a collection of conjectures. Nearly twenty years ago I discovered McCloskey's (1985) discussion of intellectual life as a conversation that links the present to both the past and the future. This column was prepared under the premise that there are too few opportunities for informal conversations about ideas that influence how we can or should use economics. I was not sure how to start such a process when I began work on the first in this series. It seemed natural to me to select articles that got me "stirred up," and write about them. As a result, a few caveats may be in order to reinforce what I said at the outset. First, it may seem I have singled out certain areas and levied some harsh criticism. That is not my intention. The "conversations" this new research has stimulated, both within and outside journals, are exciting. I wanted to offer another interpretation in order to stimulate even more conversations, hopefully within environmental economics.

Second, I may well have oversimplified or misrepresented some arguments. I apologize in advance. Where I have messed up, tell me. This is what a conversation is about, and it is how we all learn. Finally, I cannot hope to be representative. When I miss things you think are important, please tell me.

What's next? In an early draft of this first essay, I included some hints, but my bosses needed to save some space and told me to delete them all. Instead, they advised: "keep 'em guessing." So you will have to wait for Reflections No. 2.

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Announcements

New AERE Fellows

The Association of Environmental and Resource Economists (AERE) announced six new AERE Fellows at its annual meeting, held in conjunction with the Allied Social Science Associations meeting in Chicago, Illinois, in January 2007: Richard C. Bishop, University of Wisconsin, Madison; Nancy E. Bockstael, University of Maryland, College Park; Ronald G. Cummings, Georgia State University; Anthony C. Fisher, University of California, Berkeley; Geoffrey M. Heal, Columbia University; and Clifford S. Russell, Vanderbilt University.

AERE Publication of Enduring Quality Award

AERE conferred this annual award at the AERE meeting in Chicago in January 2007, to H. Scott Gordon, "The Economic Theory of a Common-Property Resource: The Fishery," *Journal of Political Economy* 62, 2 (April 1954): 124–142; and Anthony Scott, "The Fishery: The Objectives of Sole Ownership," *Journal of Political Economy* 63, 2 (April 1955): 116–124. H. Scott Gordon is a distinguished professor emeritus at Indiana University. Anthony Scott is a professor emeritus at the University of British Columbia.

Petry Research Prize for the Economics of Climate Change

Richard Newell and William Pizer were presented with this award by AERE at

the Chicago meeting in January 2007. The award was announced in July 2006 at the World Congress of Environmental and Resource Economists in Kyoto, Japan. The award recognizes their paper "Discounting the Distant Future: How Much Do Uncertain Rates Increase Valuations?" published in the *Journal of Environmental Economics and Management* in July, 2003.

Officers and Board Members of AERE for 2007

The 2007 slate of AERE Officers is: President, Trudy Ann Cameron, University of Oregon; Past President, Richard T. Carson, University of California, San Diego; Vice President, Stephen Polasky, University of Minnesota; Secretary, Ann Wolverton, U.S. Environmental Protection Agency; and Treasurer, Ian W. H. Parry, Resources for the Future (RFF). The members of the Board of Directors and their terms of office are as follows: J. R. DeShazo, University of California, Berkeley (2006–2008); Lawrence Goulder, Stanford University (2007–2009); Gloria Helfand, University of Michigan (2006–2008); Alan Krupnick, RFF (2005–2007); Carol McAusland, University of Maryland, College Park (2007–2009); Marca Weinberg, U.S. Department of Agriculture (2005–2007); plus ex-officio members Marilyn M. Voigt (executive director) Charles F. Mason (managing editor of the *Journal of Environmental Economics and Management*) and Robert N. Stavins (editor of the *Review of Environmental Economics and Policy*).

Call for Papers

The 2008 winter meeting of the ASSA will be held in New Orleans, Louisiana, on January 4–6, 2008. Authors wishing to have a paper considered for the AERE sessions should send a two- to three-page PDF file by March 31, 2007. Submission details can be found on the AERE web site at <http://www.aere.org>.

Fifteenth EAERE Annual Conference

The European Association of Environmental and Resource Economists (EAERE) will hold its annual conference at the University of Macedonia in Thessaloniki, Greece, June 27–30, 2007. Further information about the conference is available at <http://www.eaere2007.gr>.

European Summer School in Resource and Environmental Economics

EAERE, the Fondazione Eni Enrico Mattei, and the Venice International University will sponsor their annual summer school in resource and environmental economics for Ph.D students in Venice, July 4–10, 2007. This year's topic is Trade, Property Rights, and Biodiversity. Further information about this year's summer school is available at <http://www.feem.it/ess07>.

Request for Nominations

On behalf of EAERE and the Kempe Foundation, nominations are invited for the 2007 Eric Kempe Award in Environmental and Resource Economics. The ten-thousand-euro award is for the best paper in the field of environmental and resource economics published in a refereed journal in

2004–2006. At least one of the paper's authors must be affiliated with a European research institution. The deadline for nominations is March 30, 2007. Complete submission information is available at <http://www.eaere.org/ek.html>.

Call for Nominations for EAERE President and Council Members

EAERE invites nominations for president and council members, with voting to take place in the fall of 2007. The deadline for nominations is April 30, 2007. Further information is available at <http://www.eaere.org/elections07.html>.

Call for Papers

The National Bureau of Economic Research (NBER) Summer Institute Workshop on Environmental Policy will be held in Cambridge, Massachusetts, on July 23–24, 2007. The NBER provides continental breakfast and lunch each day, as well as limited support services. The Environmental Workshop has funding from the U.S. Environmental Protection Agency to pay for travel cost of one author per accepted paper. Economists at the assistant professor level or above who are interested in participating in the 2007 Summer Institute should submit a curriculum vitae and a one-page abstract of any paper they would like to present to Summer Institute, National Bureau of Economic Research, 1050 Massachusetts Avenue, Cambridge, MA 02138. Submissions can also be sent via e-mail to confer@nber.org in either PDF, Word, or WordPerfect format. The deadline for receipt of submissions is March 19, 2007, or until selection decisions are made, whichever is later.

Papers and Videos Available from Conference

RFF hosted a conference on Frontiers of Environmental Economics on February 26–27, 2007, in Washington, D.C. The conference included nine commissioned papers drawn from an international competition with 175 proposals, plus a panel discussion on frontiers of environmental economics. The commissioned papers and videos of the paper presentations, discussants' comments, and panel discussion are available at <http://www.rff.org/rff/Events/Frontiers-of-Environmental-Economics.cfm>.

New Appointments

Richard G. Newell, formerly a senior fellow, Resources for the Future, Washington, D.C., is the Gendell Associate Professor of Energy and Environmental Economics, Nicholas School of the Environment and Earth Sciences, Duke University, Durham, NC, effective January 2007.

Alexander Pfaff, Columbia University, New York, NY, becomes an associate professor, Department of Public Policy Studies, Duke University, Durham, NC, effective July 2007.

V. Kerry Smith, formerly a University Distinguished Professor, North Carolina State University, Raleigh, NC, is W. P. Carey Professor of Economics, W. P. Carey School of Business, Arizona State University, Tempe, AZ.

Jeffrey Vincent, a professor in the Graduate School of International Relations and Pacific Studies, University of California, San

Diego, will become the Korstian Professor of Forest Economics and Management, Nicholas School of the Environment and Earth Sciences, Duke University, Durham, NC, effective July 1, 2007.

W. Kip Viscusi, formerly the John F. Cogan, Jr., Professor of Law and Economics, Harvard Law School, is a University Distinguished Professor of Law, Economics, and Management, Vanderbilt University, Nashville, TN.

Instructions for Submission of Announcements

When submitting information for inclusion in the Announcements, please observe the following guidelines. Calls for papers, notices of professional meetings, and other announcements of interest to environmental economists should be submitted in one paragraph that contains all relevant information. Use the examples above as a style guide.

News of individuals should be labeled as to category: (1) retirements; (2) deaths; (3) new appointments; (4) promotions; (5) important administrative appointments; (6) leaves of absence for special appointments; (7) resignations; and (8) miscellaneous. Please provide the individual's full name, present place of employment, new title, new institution and date on which change is to occur.

The deadline for the winter 2008 issue is December 1, 2007. Please submit all information via e-mail to the *Review's* editorial office at reep@aere.org. The editors reserve the right to edit material received.