Abstract and Keywords

This chapter begins with a brief general overview of the economics of environmental law. It then focuses on recent developments in the field of environmental law and economics, with an emphasis on the experience of the United States. When setting environmental policy, decision makers must address two general types of questions. The first concerns the ends of environmental policy, and examines the socially desirable level of environmental quality. The second type of question concerns the means of policy making and focuses on the types of regulatory instruments that will be used and the allocation of responsibility between governmental actors. Section 2 addresses the first type of question concerning the goals of environmental policy. Sections 3 and 4 address the means of environmental policy, focusing on instrument choice and jurisdictional allocation, respectively.

Keywords: economics, environmental law, environmental policy, instrument choice, jurisdiction

20.1. Introduction

The law and economics perspective provides a useful lens for many environmental policy questions. Normative deliberation concerning the construction of environmental policy can be informed by an economics perspective. Economic analysis can also be brought to bear on empirical questions concerning the effects of environmental policies (Faure 2012) and the political economy factors that affect the selection of environmental policies.
Environmental Law and Economics

(Keohane, Revesz, Stavins 1996; Burtraw 2012). Indeed, the overlap between environmental policy and economics is sufficiently extensive that environmental economics constitutes a distinct discipline within the field of economics (Field and Field 2012).

Revesz and Stavins (2007) presented an overview of environmental law and economics, focusing on three themes that have been of particular importance in the field: cost–benefit analysis, instrument choice, and the allocation of policymaking responsibility in a federal system. That work canvassed the robust literature that had developed on those themes and examined the political and policy contexts that affected the application of law and economics to environmental policymaking.

There have been important developments since the publication of Revesz and Stavins (2007). Climate change has become an even more dominant issue in environmental law, both in the United States and internationally. The scale of the threatened harms from climate disruption, the global and diffused nature of greenhouse gas emissions, the degree of scientific uncertainty, the long time horizons involved, and the extent of economic transformation needed to substantially curtail emissions all make climate change a uniquely vexing environmental problem (Intergovernmental Panel on Climate Change [IPCC] 2013).

The issue of climate change has precipitated important changes in the field of environmental law and economics. Many normative and positive challenges are posed by climate change and attempts to reduce greenhouse gas emissions, and scholars have responded to those challenges with important conceptual and empirical advances. The political context of environmental law and economics has also been affected by conflict over climate policies. After providing a brief general overview of the economics of environmental law, this chapter will focus on these recent developments, with an emphasis on the experience of the United States. Consistent with Revesz and Stavins (2007), this chapter focuses on pollution control and does not examine natural resource management.

When setting environmental policy, decision makers must address two general types of questions. The first concerns the ends of environmental policy, and examines the socially desirable level of environmental quality. The second type of question concerns the means of policymakers and it focuses on the types of regulatory instruments that will be used and the allocation of responsibility between governmental actors. Section 20.2 addresses the first type of question concerning the goals of environmental policy. Sections 20.3 and 20.4 address the means of environmental policy, focusing on instrument choice and jurisdictional allocation, respectively.
20.2. Cost-Benefit Analysis

There is widespread agreement that environmental quality is a good worth providing. Not only do people enjoy the benefits of environmental quality directly—for example, through improved health and better recreational opportunities—but natural systems provide the foundation for many diverse forms of economic activity (such as agriculture) and, indeed, create the basic conditions for human life to exist at all. But although it is impossible to dismiss the value of a clean environment, there is much less agreement on the level of environmental quality that society ought to pursue. The question of how clean is clean enough? is an essential preliminary to environmental policymaking, and one on which economics provides useful guidance.

20.2.1. Normative Issues and Analysis

Welfare economics provide a general framework for answering questions concerning the ends of environmental policy. The general recommendation is that environmental policy should be selected to maximize well-being, measured by the value that individuals place on improved environmental quality minus the value of sacrifices that must be made to achieve those improvements. Because most environmental policy involves tradeoffs between positive and negative consequences, the value of those consequences must be weighed against each other. The dominant technique for weighing policy consequences and estimating the net effects of policy options is cost–benefit analysis.

20.2.1.1. General Framework

Pollution presents a classic market failure that can be addressed through appropriate government intervention. Pollution can be described as an externality (Pigou 1920) in which the “activity of one agent ... affects the well-being of another agent and occurs outside the market mechanism” (National Research Council 2010). The solution proposed by Pigou is a tax or fee to internalize the external costs imposed by pollution. Such a move is not necessary in world of well-defined property rights and low transaction costs because private bargaining will account for all relevant effects (Coase 1960; Krutilla and Krause 2011). Nevertheless, the existence of transaction costs and incomplete property rights imply that private bargaining alone will not always result in a social-welfare-maximizing outcome.

Once it is determined that government intervention can, in principle, improve well-being, the question of the socially desirable level of pollution control must be addressed. Within the field of welfare economics, the traditional answer is the test developed by Kaldor (1939) and Hicks (1939) of potential Pareto improvements. A Pareto improvement is one that makes at least one person better off and no one worse off (Pareto 1896). A
potentially Pareto improvement is one in which the beneficiaries of the policy could fully compensate those who are burdened by the policy.

Kaldor–Hicks efficiency is the basis for formal cost–benefit analysis (Zerbe and Bellas 2006). A measure is Kaldor–Hicks efficient if it maximizes the difference between the value of its benefits (as measured by the beneficiaries) and its costs (as measured by bearers of those costs). Because total benefits typically increase at a decreasing rate while cost increases at an increasing rate, net benefits are maximized by policies that equate marginal benefits with marginal costs.

There is a robust debate over the normative attractiveness of cost–benefit analysis as a means of identifying socially desirable environmental policy. Issues that are relevant to this debate include the normative status of preference satisfaction (Adler and Posner 2006), the importance of distribution of costs and benefits for well-being (Adler 2012; Cai et al. 2010; Farrow 2011), and broader critiques concerning incommensurability (Anderson 1995; Ackerman and Heinzerling 2004) or commodification (Radin 2001). While many of the criticisms leveled against cost–benefit analysis apply generally to normative economics, the use of cost–benefit analysis to evaluate environmental policies has been particularly controversial (Kysar 2010). Additional conceptual frameworks for evaluating environmental policy include environmental rights (Kelman 1981a), environmental justice (Rhodes 2003), non-anthropocentric approaches (Ariansen 1998), green growth (Livermore 2013), and sustainable development (Dasgupta and Heal 1974; Stiglitz 1974; Solow 1974; Organisation for Economic Cooperation and Development [OECD] 2004). Many of these concepts can be understood as emphasizing distributional, rather than efficiency, considerations (Stavins et al. 2003). For example, one definition of sustainable development that is reasonably well accepted in the economics community is monotonically increasing consumption (Solow 1986; Hartwick 1977; Daly 1990), which concerns the intergenerational allocation of resources, rather than the static maximization of welfare.

Environmental policymaking poses several methodological difficulties for cost–benefit analysis. Several of these are described in detail in Revesz and Stavins (2007). The most important environment-specific challenge is to provide valid monetary estimates of the value of hard-to-price goods, including non-market goods (like clean air) (Alberini and Scarpa 2005; Banzhaf 2010; Phaneuf et al. 2009) and mortality risk reduction (Cropper et al. 2011). Continent valuation (e.g., stated preference studies) remains the most widespread tool to estimate non-market values (Viscusi et al. 2008). While this approach has been endorsed by leading economists, it remains controversial (Hausman 2012). In recent years, the concept of ecosystem services—the goods and services provided by natural systems—has been used to help structure research on the benefits of environmental protection (Goulder and Kennedy 2011; Brown et al. 2007). An important challenge for the ecosystems services approach is to identify estimation methods that disaggregate intermediary goods, such as water quality from final services such as recreation or human health (Keeler et al. 2012).
20.2.1.2. Discounting and Future Generations

Carbon dioxide, the most important greenhouse gas pollutant, has a long atmospheric lifetime, about 100 years (IPCC 2013). In addition, global warming processes are not instantaneous, but involve the interaction of large, complex physical systems that may involve irreversible feedback loops (IPCC 2013). The consequences of greenhouse gas emissions today, then, will be felt for many years into the future. As in many other environmental contexts, limits on greenhouse gas emissions impose immediate costs to produce benefits that will accrue only in the long term.

Traditionally, future benefits have been discounted at a constant rate. The choice of a discounting procedure and rate is particularly important in the case of climate change because of the very long time horizons involved (Revesz and Shahabian 2011). The standard approach to analyzing this problem is contained in the following formula (Arrow et al. 1996):

\[ d = \rho + \theta g \]

where \( d \) is the discount rate, \( \rho \) is the rate of pure time preference, \( \theta \) is the absolute value of the elasticity of marginal utility of consumption and \( g \) is the growth rate of per capita consumption.

There are two interpretations for the first term, \( \rho \). Under a prescriptive approach, \( \rho \) would be derived from ethical principles (Arrow et al. 1996). There is no strong consensus concerning whether a pure time preference is normatively justified. Many prominent economists agree with the proposition, first articulated by Ramsey (1928), that \( \rho \) should be zero (Broome 1992; Dasgupta 2008; Cline 2006; Harrod 1966; Heal 2009; Koopmans 1967; Philibert 1999; Solow 1974). Others disagree (Arrow 1999; Beckerman and Hepburn 2007).

Revesz (1999) and Revesz and Shahabian (2011) argue that there is an ethical distinction between intergenerational and intragenerational discounting that is relevant for determining the appropriate pure rate of time preference. The latter reflects individuals’ preferences to spread consumption across their lifetime. The former reflects a social decision concerning the allocation of resources between individuals. While there are reasons to endorse policies that reflect individuals’ desire to consume sooner rather than later, there is no straightforward justification for a bare social preference for current over future generations.

The descriptive approach avoids the preceding normative questions and simply substitutes observed market interest rates for the discount rate in cost–benefit analysis. The benefit of this approach is that it does not attempt to address difficult questions of intergenerational equity. The problem is that it fails to provide a normative reason for why market interest rates are an appropriate source for the social discount rate (Revesz and Shahabian 2011). Sophisticated defenders of intergenerational discounting recognize the possibility for “net welfare losses and distributional inequity” but argue that the issue
of moral obligations to future generations should be treated as a separate inquiry from efficiency (Sunstein and Rowell 2007).

Once society decides how to trade off utility between generations, discounting can be used to account for opportunity costs in determining the degree to which pollution control should be included in the basket of future-oriented investments (Schelling 1995; Samida and Weisbach 2007; Weisbach and Sunstein 2009).

Accounting for the marginal utility of consumption through growth discounting also generates difficulties. As a preliminary matter, long-term growth is difficult to predict (Moyer et al. 2013). The countries most likely to benefit in the future from current climate change mitigation—developing countries in the tropics—are far poorer than the industrial powers that are most likely to pay in the present for such mitigation (Ruhl 2012). And even in the relatively distant future, these developing countries may be poorer than developed countries are today. Scholars have also argued that greenhouse gas reductions in developed countries can be viewed as foreign aid and noted that intragenerational wealth transfers are a more efficient mechanism to reduce global inequality than climate mitigation (Schelling 1995; Posner and Sunstein 2007).

There appears to be an emerging consensus that discount rates should decline with the length of the time horizon, with higher discount rates applied to the near-term future and lower discount rates applied to the long-term future (Gollier and Weitzman 2010). A declining discount rate approach has been adopted by France and the United Kingdom (Cropper 2012). The first argument in favor of such an approach is that it is more consistent with observed behavior. Stated preference studies typically indicate that individuals apply a very low discount rate to benefits to future generations (Cropper et al. 1992; Johannesson and Johansson 1997). Even in individual market behavior, hyperbolic discounting is commonly observed (Laibson 1997), although it raises rationality concerns (Skog 2005).

\[\text{(p. 514)} \]

The second and more powerful argument in favor of declining discount rates stems from uncertainty about the appropriate discount rates (Arrow et al. 2013; Gollier and Weitzman 2010). When these rates are uncertain, the utility maximizing approach requires that decisions be made based on expected discounted costs and benefits, not discounted expected costs and benefits (Weitzman 1998, 2001). The consequence is that the correct discount rate is lower than the mean of the probability distribution of possible rates. Over sufficiently long time horizons, the lowest rate in the distribution dominates (Weitzman 2001). Newell and Pizer (2003) show that rational treatment of interest rate uncertainty, based on historic variance in US Treasury bill rates, leads to quickly declining discount rates.

\[\text{20.2.1.3. Employment Effects} \]

A growing area of research interest, spurred in part by changes in the political context of cost–benefit analysis discussed below, concerns the effects of environmental regulation
Environmental Law and Economics

on employment. A threshold question is whether, and how, effects on labor markets ought to be accounted for in cost–benefit analyses of environmental regulation.

The standard approach for government agencies in the past had been to assume a well-functioning, full-employment labor market (Environmental Protection Agency [EPA] 2000). Under these conditions, workers hired to comply with an environmental regulation are reallocated from positions with similar wages, and workers that are laid off because an environmental regulation find new positions at similar compensation levels.

When the assumption of full employment is relaxed, the estimation of the labor effects of environmental regulation is extremely difficult (Coglianese et al. 2014). Morgenstern et al. (2002) note three distinct potential effects of regulation on employment. A demand effect occurs if regulation increases production costs, thereby increasing prices and lowering the quantity of production, resulting in lower demand for labor. A cost-effect occurs if plants must add more capital and labor per unit of output, potentially increasing demand for labor. A factor-shift effect occurs if a regulation induces a shift in spending from capital to labor or vice versa. The authors note that the net effects of these three effects are ambiguous, and in a study of four highly polluting and regulated industries, find mild positive employment effects from regulation. Sectoral studies have found negative employment effects from environmental regulation, but may measure employment shifts rather than net losses (Greenstone 2002; Kahn 1997). Walker (2011) examines wage effects from the Clean Air Act, finding that wages within regulated industries declined because of environmental controls. Office of Management and Budget (OMB) (2011) provides a useful overview of research on employment effects from environmental regulation.

Masur and Posner (2012) argue that, when realistic economic conditions are taken into account, there can be substantial costs associated with layoffs. Not only do laid-off workers incur transition costs, including job search costs and time away from work, but there are psychological and physical hardships associated with joblessness that are not fully captured by lost wages or search costs. Adler (2014) discusses how a welfare-oriented approach would account for these types of harms.

Labor effects from environmental regulation are not always negative. If firms that must hire new workers to comply with an environmental regulation draw from a pool of unemployed labor, then the social costs are below the wages that are paid out to those workers, because the opportunity costs of these workers are very low (Bartik 2013). There may also be physical and psychological benefits associated with hiring unemployed workers that imply a negative social cost (CBO 2012).

20.2.1.4. Behavioral Economics and the Energy Paradox

Since the mid-1990s, scholars have begun applying the insights of behavioral economics to the law (Krieger 1995; Korobkin and Ulen 2000; McAdams 1997; Sunstein 1996). The behavioral revolution has touched environmental economics as well (Shogren and Taylor 2008). Arguably, the most important consequence of behavioral research for cost–benefit
analysis of environmental policy concerns the issue of energy efficiency standards. Measures to promote energy efficiency have taken on increasing importance as a low-cost means of reducing greenhouse gas emissions.

Energy-efficiency standards have two potential categories of benefits. By reducing energy consumption, they can reduce externalities such as air pollution from burning fossil fuels or thermal pollution associated with nuclear power generation. A second category of benefits is consumer savings. Consumer savings, however, are complicated by the fact that rational consumers should be willing to invest in all net present value-positive energy-efficiency improvements (Jaffe et al. 2004). Therefore, under the standard model of rational economic behavior, it should be impossible for government regulation to generate consumer benefits through energy-efficiency mandates in most circumstances.

Notwithstanding this prediction, there is much literature demonstrating widespread underinvestment in energy efficiency (Gillingham et al. 2009). This phenomenon is referred to as the “energy paradox” in the environmental economics literature (Jaffe and Stavins 1994). Several behavioral hypotheses have been forwarded to explain the energy paradox: consumers may myopically apply above market discount rates to future energy savings (Alcott and Wozny 2012); account for energy efficiency only after having settled on other product characteristics (Geistfeld et al. 1977); demonstrate loss aversion (Greene 2008); incorrectly associate energy efficiency with poor performance (Rogers and Shrum 2012); or use other decision heuristics that fail to account for efficiency (Helfand and Wolverton 2011). There is no consensus on the dominant source of the energy paradox.

Despite the lack of a strong theoretical understanding of the energy paradox, its persistence in the marketplace indicates that consumer savings are a valid benefit associated with energy-efficiency standards. While market-based tools, such as an energy tax, are likely to be lower-cost tools for achieving emissions reductions (Karplus and Paltsev 2012), energy-efficiency standards may be justified even in situations where they have no environmental benefits, for example when implemented together with a comprehensive cap on emissions. In those cases, efficiency standards could be justified purely based on consumer savings (Bubb and Pildes 2014).

### 20.2.2. Positive Issues and Analysis

Cost–benefit analysis has become the dominant paradigm for environmental policy analysis in the United States, and its use is now becoming more common around the globe. But that does not mean that the technique is uncontroversial, and debates about its use continue.
20.2.2.1. A Changing Political Context for Cost–Benefit Analysis

In US environmental law, two general alternatives to cost–benefit analysis for setting environmental standards are absolute standards (Sinden 2005) and feasibility standards (Driesen 2005). Absolute standards seek to set environmental quality at levels that present zero (or negligible) risk of harm. The National Ambient Air Quality Standards (NAAQS) under the Clean Air Act, which must be set at a level “requisite to protect the public health” with an “adequate margin of safety” are the most important examples of an absolute health-based standard in US environmental law. Feasibility standards are set to maximize stringency, subject to the constraints of “physically impossible environmental improvements” or standards “so costly that they cause widespread plant shutdowns” (Dreisen 2011). Both absolute standards and feasibility standards are subject to serious objections. Absolute standards raise conceptual and practical problems if there is no known threshold that presents zero risk (McGarity 1979; Coglianese and Marchant 2004; Sunstein 1999). Furthermore, and paradoxically, the majority of recent NAAQS (adopted under an absolute health-based approach) are less stringent than would be economically efficient, which undermines the normative justification for such an approach (Livermore and Revesz 2014). Feasibility standards have been criticized as “creating significant problems of over- and underregulation” because they are completely insensitive to some costs while weighing other costs as infinitely high (Masur and Posner 2010).

Perhaps in part because of the limitations of these alternatives, cost–benefit analysis has come to be the dominant framework for US environmental policy (Sunstein 2002). While cost–benefit analysis is required in only a few environmental statutes—notably the Toxic Substances Control Act; the Federal Insecticide, Fungicide, and Rodenticide Act; and the Safe Drinking Water Act—for more than 30 years, US presidents have required that significant proposed regulations be subject to cost–benefit analysis (Revesz and Livermore 2008). This presidential requirement applies to all agency decisions unless precluded by statute. The most important case limiting the use of cost–benefit analysis within the environmental arena is American Trucking Associations, Inc v Whitman, in which the Supreme Court held that the statutory silence in Section 109 of the Clean Air Act should be interpreted to bar the consideration of costs (and therefore the use of cost–benefit analysis) for setting the NAAQS. The effect of American Trucking on other statutory provisions was called into question in the 2009 decision Entergy Corp. v Riverkeeper, Inc. In that case, the Supreme Court declined to extend American Trucking to cover a provision of the Clean Water Act that was silent on the use of cost–benefit analysis. After Riverkeeper, it appears that, except in perhaps a handful of instances, Executive Order 13,563 will apply and require agencies to justify their significant environmental regulations by reference to cost–benefit analysis.

Cost–benefit analysis and the closely related practice of regulatory impact analysis are now common practices outside the United States as well. The European Union has undertaken several important steps to promote assessment of costs and benefits of regulatory policy (Wiener 2006). Cost–benefit analyses carried out by governments,
independent academics, or non-governmental organizations is now commonly used in many developing and emerging countries to evaluate environmental policy as well (Livermore and Revesz 2013).

The global growth of cost–benefit analysis, and its universal support by presidents of both major US political parties over the past three decades, masks a complex political struggle over the degree to which concerns over economic efficiency ought to inform how environmental policy is set. Because there are important policy consequences associated with cost–benefit analysis, its use is not only debated by scholars, but by organized interest groups as well (Revesz and Livermore 2008).

In the United States, when the Reagan executive order placed cost–benefit analysis at the center of the regulatory state, protection-oriented groups (such as environmentalists) strongly opposed the move, while interest groups that favored more lax regulation (such as industry trade associations) promoted expanded use of the technique. This interest group dynamic remained remarkably consistent over time, surviving multiple changes in party control of the White House (Revesz and Livermore 2008).

Recently, however, the political alignment around cost–benefit analysis has become less stable. In light of congressional failure to enact climate legislation in 2009, the Obama administration has pursued an aggressive agenda of clean air protections, which either directly limit greenhouse gas emissions or have substantial climate co-benefits. At the same time, the administration has continued the long-standing presidential support for cost–benefit analysis and relied heavily on economic arguments to support environmental standards that are more stringent.

Three environmental rules adopted by the EPA under the Obama administration were supported by particularly persuasive cost–benefit analyses: the updated Corporate Average Fuel Economy (CAFE) Standards released in 2012; the 2013 Mercury and Air Toxics Standards (MATS); and EPA’s Cross-State Air Pollution Rule. The CAFE Standards will raise average fuel economy to 54.5 mpg by 2025, with estimated net benefits of approximately $700 billion by 2025 (National Highway Traffic Safety Administration 2012). The MATS extended air-quality standards for mercury and other toxic pollutants to power plants, with estimated net benefits of at least $27 billion in 2016 (EPA 2011a). The Cross-State Air Pollution Rule (CSAPR) is the EPA’s latest attempt to address air-quality problems presented by interstate externalities. The agency’s impact assessment estimated that the rule will produce at least $120 billion annually in net benefits by 2014 (EPA 2011b). Statutorily mandated retrospective analysis prepared by the EPA of air-quality rules adopted pursuant to the Clean Air Act Amendments of 1990 estimated that by 2020, the rules would have at least $1 trillion in net benefits and, under more favorable assumptions, up to $35 trillion (EPA 2011c).

Perhaps unsurprisingly, willingness to endorse cost–benefit analysis has shifted in light of these developments. Protection-oriented groups have shown greater openness to cost–benefit analysis (Livermore and Revesz 2011). At the same time, regulatory skeptics have distanced themselves from the technique (Volokh 2011) as did the 2012 Republican
presidential nominee Mitt Romney, who argued that cost–benefit analysis “tend[s] to be vulnerable to manipulation and also disconnected from the central issue confronting our country today, namely, generating economic growth and creating jobs” (Romney 2011).

An important component of this political realignment is an expanded emphasis on the employment effects of regulation by the regulated community and some politicians. The phrase “job-killing regulation,” which appeared only four times in major US newspapers in 2007, appeared nearly 700 times in the same outlets 4 years later (Livermore and Schwartz 2014). During the 112th congressional session, employment effects were used to justify several bills to reduce EPA’s regulatory authority. The House Republican Plan for America’s Job Creators included several provisions affecting the rulemaking process (House Republican Conference 2011) and the Regulatory Freeze for Jobs Act, which would block all new significant regulations until unemployment dropped below 6%, passed the House in July 2012.

Estimates of the employment effects of regulation have become common in political discourse over environmental policy (Livermore and Schwartz 2014). These estimates, which are typically based on input–output or computable general equilibrium models, are highly sensitive to analytic assumptions and modeling choices, resulting in widely disparate results. In one telling example, a report by the America Coalition for Clean Coal Electricity estimated that two EPA rules would trigger 1.4 million job losses, while a Political Economy Research Institute study predicted the same two rules would spur a 1.4 million job gain (Livermore and Schwartz 2014).

Agencies have also begun to include employment effects alongside cost–benefit analyses. The EPA in particular has included a statement on employment effects in several high profile rulemakings (e.g., EPA 2012a; EPA 2012b). Employment effect estimates generated by agencies are often quite modest, providing a useful counterweight to extravagant jobs claims made by advocacy organizations.

20.2.2.2. Social Cost of Carbon

The application of cost–benefit analysis to regulations with greenhouse gas emissions has become a particular focal point for political conflict over the methodology. In 2009, the Obama administration convened an interagency task force that was charged with developing a “social cost of carbon” to be used across the government to assign a monetary value to greenhouse gas emission reductions in cost–benefit analyses of rulemakings (Interagency Working Group on Social Cost of Carbon [IWG] 2010). The original central estimate provided by the 2010 report for Year 2015 was $24 (2007 dollars), using a 3% discount rate (IWG 2010), and an updated estimate was released in 2013 of $38 (IWG 2013). Since being developed, the social cost of carbon has been used in a number of important rulemakings, including EPA’s fuel-efficiency standards and several Department of Energy appliance efficiency rules (e.g., EPA 2010a; Department of Energy 2013).
The taskforce based its estimate on three Integrated Assessment Models (IAMs) that link the physical and economic effects of climate change: the Dynamic Integrated Model of Climate and the Economy (DICE) (Nordhaus and Sztorc 2013); the Climate Framework for Uncertainty, Negotiation, and Distribution (FUND) model (Anthoff and Tol 2013); and the Policy Analysis of the Greenhouse Effect (PAGE) model (Hope 2013). These models translate predictions concerning greenhouse gas emission, temperature change, and physical impacts associated with climate change (such as sea-level rise or effects on agricultural productivity) into monetary terms.

Because many of the systems that are represented by these IAMs are not well understood, there is considerable uncertainty concerning the accuracy of their estimates (Pindyck 2013). Some scholars have argued that these uncertainties make climate change a special case where cost–benefit analysis cannot be usefully applied (Masur and Posner 2011; Rose-Ackerman 2011). These arguments have been repeated by industrial trade associations and politicians that oppose greenhouse gas regulations in support of efforts to prohibit the use of the social cost of carbon. Environmental advocacy groups, on the other hand, have argued that the social cost of carbon can be a useful tool for setting climate policy, although they argue that current estimates are too low, because, for example, they fail to adequately account for catastrophic outcomes (Environmental Defense Fund et al. 2013; Weitzman 2009; Weitzman 2011). The social cost of carbon, then, represents another example where the traditional interest group alignment concerning cost–benefit analysis has been inverted by new political dynamics.

20.3. Instrument Choice

Identifying the socially desirable level of environmental quality is useful only if there are means to achieve that end. This section explores different types of government interventions (i.e., policy instruments) that can be used to reduce pollution. It first discusses normative questions and then examines instrument choices that have been made in US environmental policymaking. Section 20.4 will explore questions concerning the level of government that is best suited to implement these instruments.

20.3.1. Normative Issues and Analysis

There are a wide range of policies that can be adopted to achieve environmental goals, including labeling and disclosure requirements, technology standards, liability regimes, effluent fees, and tradable emission allowances. These policies may differ along a number of important dimensions, including the flexibility they provide for regulated actors, their ease of enforcement, the information necessary to implement the policy, and the incentives they provide for technological development on the part of private actors.
(Goulder and Parry 2008). Selecting among instruments involves a sometimes-complex inquiry that is conceptual, as well as practical, and context specific.

20.3.1.1. General Framework

For a given environmental goal, the cost-effective policy will be the one that achieves that goal at the lowest total cost (Office of Management and Budget 2003). While there is substantial controversy about appropriate environmental goals, the aspiration of cost-effectiveness is widely shared. In general, cost-effective policies will equalize marginal abatement costs across all pollution sources or which reductions are possible. When marginal abatement costs are not equal, lower-cost abatement opportunities have not been fully implemented, implying that cost-effectiveness has not been achieved.

An important qualification to the general principle that marginal costs be equalized is the need to ensure compliance with environmental requirements. Enforcement may be easier at some sources than others: point sources of pollution are easier to monitor than non-point sources; inspection of a small number of large sources is less costly than inspection of a large number of small sources; certain companies or private individuals may be judgment proof, undermining the incentive effect of potential penalties (Farmer 2007). For these reasons, and others, the aspiration of equalizing marginal abatement costs may sometimes be subject to practical enforcement constraints. Cost-effective policies will achieve environmental goals at the lowest aggregate costs, including monitoring, inspection, and enforcement, meaning that some theoretical lower-cost abatement opportunities may not be realized.

Dynamic effects may also provide reasons to depart from cost-effectiveness in the short term. Environmental regulations often generate incentives for the development and deployment of new technologies (Downing and White 1986; Ellerman et al. 2003; Malueg 1989; Milliman and Prince 1989). Frequently, cost-effective instruments will provide the correct incentive for efficient technological development: firms will make marginal decisions to invest in new technologies or use existing technologies in ways that minimize the present value of their compliance costs (Zerbe 1970; Downing and White 1986; Milliman and Prince 1989). Positive externalities associated with knowledge spillovers, however, may result in underinvestment in technological development (Katsoulacos and Xepapadeas 1996). In such cases, regulations designed to require firms to overinvest in new technologies (relative to their private costs and benefits) may be welfare maximizing (Jaffe et al. 2001). In the long run, this approach would be cost-effective, but in the short term, lower-cost abatement opportunities may be forgone.

An additional element of instrument choice concerns the selection of regulatory tools in the face of uncertainty. A social decision maker can be uncertain about either environmental damages or abatement costs, or both. Weitzman (1974) shows that when a regulator must either set a price (e.g., through effluent fee) or a quantity (e.g., through a pollution cap) cost uncertainty can significantly affect this choice, with the relative elasticity in damage and cost functions determining whether a price instrument is more efficient that a quantity instrument. Recent research shows that nonlinear taxes, graded
quantities, or hybrid tax-quantity instruments have advantages over either flat prices or fixed quantities (Roberts and Spence 1976; Weitzman 1978; Kaplow and Shavell 2002; Pizer 2002; Fell et al. 2012).

20.3.1.2. Market Mechanisms versus Command and Control Regulation

A particularly important choice that is often presented in the environmental context is between command-and-control style regulation and market-based approaches. The typical example of a command-and-control environmental regulation is a design standard that requires a specific pollution reduction technology to be adopted by all regulated firms. An example of a market-based mechanism is a comprehensive, economy-wide “cap-and-trade” system of tradable emission allowances. There are many alternatives along a command-to-market continuum (Freeman and Kolstad 2006). Performance-based standards, which set effluent limits but do not specify a particular technology, are more market-like than they are a prescriptive design standard (EPA 2010b). Flexibility can be added to a command-and-control standard by allowing firms to comply with emissions requirement through offsets, in which new emissions must be accompanied by equivalent reductions. A particular version of an offset mechanism is the use of “bubbling” to treating a facility or firm as a single source. The use of bubbling under the Clean Air Act new source performance standards provision was the substantive question at the heart of *Chevron v NRDC*, the case that sets out the contemporary standard of judicial deference to agency statutory interpretations. Compliance flexibility can also be enhanced through banking and/or borrowing, in which past or future emissions reductions can “count” toward emissions limits. Environmental liability rules (Viscusi and Zeckhauser 2011) and labeling and disclosure can be used to achieve environmental goals (Thaler and Sunstein 2008) in place of, or as a supplement to, ex ante controls.

More market-like instruments are often favored on cost-effectiveness grounds because they provide firms with flexibility in achieving emissions reductions and tend to equalize marginal abatement costs across firms (Montgomery 1972; Baumol and Oates 1988; Tietenberg 1995). On the other hand, command-and-control mechanisms may be easier to enforce (Grossman and Cole 1999) and they avoid problems that arise when emissions are not spatially or temporally fungible (Sado et al. 2010).

20.3.1.3. Environmental Transitions and Grandfathering

As in all areas of policy change, environmental law creates issues of retroactivity when private or public actions are undertaken under a prior regime that can, at least potentially, be governed by the new regime (Fisch 1997). The issue of retroactivity can be especially important in the environmental context because environmental standards can affect decisions that are extremely widespread (e.g., car purchase decisions in the context of fuel economy standards for automobiles) or touch on extremely high-value infrastructure investments (e.g., power plants affected by air-quality standards). Many
environmental regimes adopt a bifurcated approach in response to the retroactivity problem, treating new emissions sources differently than existing sources.

Three general justifications can be given for a bifurcated approach. First, existing sources have a life cycle of depreciation and technological obsolescence for reasons unrelated to the environmental regime. Requiring expensive investments to upgrade existing emissions sources that will soon be retired may be unjustified. Second, the marginal cost of pollution control achieved by retrofitting existing sources may be much higher than the same pollution reductions at new sources. Finally, public choice theory may predict that existing sources may act as an effective lobbying coalition against economically justified environmental protections; grandfathering may be the second-best solution to overcome opposition to new controls (Revesz and Westfahl Kong 2011).

An important dynamic effect of the bifurcated approach is referred to as the “old plant effect” (Ackerman and Hassler 1980). When additional costs are imposed on new plants in the form of stringent environmental standards, the life of the existing plants is extended and their replacement by new plants is delayed. The favorable treatment extended to existing sources under the bifurcated approach increases their value, and crowds out new sources based on technologies that are more efficient and would out-compete existing sources in the absence of the environmental policy. The net result on emissions from the imposition of an environmental standard when it is accompanied by an exemption for existing sources is ambiguous (Nash and Revesz 2007). The old plant effect is compounded by a public choice dynamic in which beneficiaries of a grandfathering policy lobby to extend their favorable treatment for as long as possible (Revesz and Westfahl Kong 2011).

Auction-based cap-and-trade or effluent fee systems represent an efficient approach to allocating abatement costs and investments between new and existing sources (Montero 2008). In cases where pure market-based mechanisms are infeasible, bifurcated treatment may be justified, despite the old plant effect. The standard approach to determining the optimal bifurcated treatment is through a two-step, sequential inquiry (Shavell 2008). The first question is the optimal stringency for new sources; the second question concerns the optimal transition rule for existing sources in light of the standards for new sources. The sequential approach, however, does not account for the old plant effect and, therefore, leads to better-than-optimal treatment of existing sources; jointly optimizing the grandfathering rule and the new source standard leads to superior results (Revesz and Westfahl Kong 2011).

20.3.2. Positive Issue and Analysis

Nearly a half-century after the United States embarked on major expansion in federal environmental law, this natural experiment has generated some useful results that can inform future policy design. In particular, experience with the sulfur dioxide trading
Experience with Market-Based Mechanisms

As noted by Schmalensee and Stavins (2010), “market-based policies ... were innovations developed by conservatives in the [Ronald] Reagan, George H. W. Bush, and George W. Bush administrations.” Given this pedigree, it is perhaps unsurprising that many environmental groups opposed marketable permit schemes (Hahn and Hester 1989). The only major environmental organization that showed a strong interest in developing market-based solutions to environmental problems, the Environmental Defense Fund, was strongly criticized (Krupp 2008).

Developments in the late 1980s and early 1990s shifted this dynamic, holding out the promise that a new political consensus in support of market mechanisms was attainable. The passage of the Clean Air Act Amendments of 1990, which established a national market for trading in sulfur dioxide emissions, is generally regarded as the first large-scale collaboration across the political spectrum to support a marketable permit scheme (Joskow and Schmalensee 1998). The legislation was adopted with wide bipartisan support; only five Democratic and five Republican senators voted against the bill.

The sulfur dioxide emissions-trading program is widely viewed as a major success (Chan et al. 2012). Impressive emissions reductions occurred at low costs (Ellerman et al. 1997). Use of the trading mechanisms allowed firms to deploy relatively low-cost alternatives such as fuel shifting and other production process changes (Doucet and Strauss 1994). The trading program also may have led to technological change that would not have been induced through a design standard (Burtraw 1996; Ellerman and Montero 1998; Bohi and Burtraw 1997; Keohane 2001). The program also resulted in higher-than-expected benefits as the severe health consequences associated with particulate matter (of which sulfur dioxide is a precursor) have become more clear (EPA 2011c).

The success of the US sulfur dioxide program spurred major interest in tradable allowance systems (Ayres 2000). Environmental issues for which tradable allowance systems have been implemented include fisheries management and water quality (Shortle and Horan 2006). Most important, a tradable allowance system is widely recognized as the preferred approach to climate change among political leaders (Stavins 2008). The European Emissions Trading System (ETS), created to fulfill obligations under the Kyoto Protocol, is the most important existing greenhouse gas emissions–allowances trading system (IPCC 2007). Despite some continuing difficulties in maintaining price stability, in part owing to an excessively generous overall cap on emissions, the European ETS has led to important emissions reductions and provided valuable lessons in market design (Brown et al. 2012). Within the United States, the political consensus around market-based approaches to greenhouse gas reductions was nearly universal, with presidential nominees of both major parties strongly supporting a cap-and-trade system and
environmentalists largely dropping their opposition to market mechanisms (Bernton 2008).

The promise of consensus, however, was short-lived. Comprehensive legislation was an early priority for the Obama administration. A bill to create a comprehensive cap-and-trade regime was adopted by the Democratic-dominated House of Representative without a single Republican vote.\(^\text{11}\) When the bill passed the House, debate shifted to the Senate, where it took on a highly partisan tenor. Regulated industry led a major lobbying and public relations push to oppose the legislation. Advocates and politicians that opposed the bill cast it as a tax on energy that would hurt consumers (Boehner 2009). The opposition campaign was ultimately successful, and the bill died in the Senate without being called to a vote (Stromberg 2010). Opposition to cap-and-trade became a centerpiece of the successful Republican effort to retake the House in 2010 (Good 2011). By the time of the 2012 presidential election, the issue had become a political litmus test for conservatism in the Republican primary (Weigel 2011). There is now a broadly shared view that legislation to create a market-based system to control greenhouse gas emissions is not politically achievable in the current political environment (Broder 2010). In the absence of legislation, the EPA moved forward with regulations under the Clean Air Act (Executive Office of the President 2013). While the statutory structure of the Act, especially Section 115, may be broad enough to accommodate a flexible, market-based approach (Chettiar and Schwartz 2009; Chang 2010), there is substantial legal uncertainty given the vagueness of the statutory language.

### 20.3.2.2. Labeling and Disclosure

While political consensus concerning market-based approaches to pollution control was short-lived, there has been renewed interest in labeling and disclosure as a lower cost and, therefore, more politically palatable means of improving environmental quality. In particular, Cass Sunstein’s tenure as the Administrator of the Office of Information and Regulatory Affairs (OIRA) at the White House Office of Management and Budget from 2009 to 2012 was marked by efforts to improve the design of labeling and disclosure regimes to maximize their effectiveness (Office of Information and Regulatory Affairs [OIRA] 2010).

A new fuel economy label adopted in 2011 required all new automobiles to “provide a clear statement about anticipated fuel savings (or costs) over a five-year period” (Sunstein 2012). The label simplifies and clarifies fuel economy information pertinent to consumer decision making, no longer “leav[ing] it to consumers to do the arithmetic needed to figure out the net economic effects of fuel economy standards on their budgets and lives” (Sunstein 2012; EPA 2011d). The design of the new label was foreshadowed in Sunstein and Thaler (2008), which argued for display of multiyear estimates of fuel costs.

Similarly, in 2009 the EPA promulgated a mandatory greenhouse gas reporting rule,\(^\text{12}\) which mimics disclosure programs like the Toxic Release Inventory (TRI) required by the Emergency Planning and Community Right to Know Act. The TRI, which requires
reporting of both storage and release of potentially hazardous chemicals, without requiring any additional action, has led to significant reductions in toxic releases into the environment (Khanna et al. 1997). The Greenhouse Gas Reporting Program (GGRP) will create a database covering 85%-90% of total US greenhouse gas emissions by requiring reporting from sources that “emit 25,000 metric tons or more of carbon dioxide equivalent per year in the United States,” excluding the agricultural sector (EPA 2013). The GGRP will increase the quantity of greenhouse gas (GHG) emission information, as well as its visibility and salience, among both GHG sources and the public (OIRA 2010; Cohen and Viscusi 2012).

20.3.2.3. Social Norms

There have also been recent efforts to use government institutions to influence social norms in an environmentally friendly direction, an approach that has received attention as a “libertarian paternalist” mechanism to promote social goals (Sunstein and Thaler 2008). One such example is an executive order that seeks to increase the visibility and salience of energy costs within the federal government through methods such as public scorecards and leadership awards (Sunstein 2011). As part of this effort, the federal government has partnered with the private sector to set joint goals for energy efficiency, leveraging its own efficiency improvements to spur improvements more broadly (e.g., Department of Energy 2013). These efforts expand on voluntary partnerships programs that have existed in the environmental area for some time (Borck and Coglianese 2009; Coglianese and Nash 2009).

20.3.2.4. Reducing the Effect of Grandfathering under the Clean Air Act

Although a bifurcated approach to new and existing sources is a common feature of US environmental law, in recent years the EPA has undertaken a number of significant measures designed to limit the scope of grandfathering. For example, the agency has engaged in a multiyear effort to establish a marketable permit scheme to control interstate pollution that reduces favorable treatment for inefficient pre-Clean Air Act sources. As discussed above, the most recent iteration of this effort, the Cross-State Air Pollution Rule adopted in 2011, was struck down by the D.C. Circuit in 2012. Also in 2011, EPA adopted the MATS, which limits the emissions of a number of toxic air pollutants from both new and existing power plants. And, in 2013, President Obama indicated that in 2014, EPA will propose a rule limiting the greenhouse gas emissions of existing power plants under Section 111(d) of the Clean Air Act. The combined effect of these approaches could significantly accelerate the shutdown of existing plants that had been grandfathered for more than 40 years.
20.4. Jurisdictional Allocation

The existence of externalities, transaction costs, and imperfect property rights imply that an unregulated marketplace is unlikely to provide efficient levels of environmental quality—government intervention is necessary. Once that preliminary observation has been made, there is a second question concerning how governmental authority over environmental policy ought to be allocated among different levels of government, from the local (municipalities) to the global (international institutions). This question has important implications for the effectiveness and legitimacy of environmental policy.

20.4.1. Normative Issues and Analysis

The allocation of policymaking responsibility across jurisdictions is an important question in many legal domains, and one that has caught the attention of law and economics scholars (Faure and Johnston 2009). Economics, in particular, can address the role of incentives in determining whether a policymaking context is amenable or not to local control. Environmental policymaking presents a number of specific issues that are relevant to questions of jurisdictional allocation.

20.4.1.1. General Framework

Several important inputs into environmental protection can vary by geographic region, a fact that increases the desirability of locally tailored standards. These geographically variable inputs include the marginal costs of pollution control, preferences concerning the value of environmental quality (compared with other goods), and the level of exposure to environmental risk (which is affected by population density, among other factors) (Mendelsohn 1986; Nordhaus 1994). To the extent that information about geographic variability is more likely to be held by local government officials, there is a justification for a rebuttable presumption in favor of local control over environmental policy (Revesz and Stavins 2007). At the same time, environmental protection is also an area where, at least in some instances, the local-control presumption is rebutted by the existence of interjurisdictional externalities (Revesz 1996).

Alternative justifications have been given for why national-level control over environmental protection is desirable, but they have important weaknesses. Most prominently, scholars have argued that environmental protection presents a “race-to-the-bottom” problem in which interjurisdictional competition leads to inefficiently low levels of regulation (Esty 1996). Basic models have demonstrated, however, that rational, self-regarding jurisdictions in a perfectly competitive market will arrive at efficient levels of pollution control (Revesz 1992). If the assumption of rationality or perfect competition is relaxed, jurisdictions may over- or underregulate (Revesz 1997). Federal floors (which are common in environmental law), then, are no better justified than federal ceilings (which
are relatively uncommon, although not unknown). In addition, to the extent that
environmental protection is federalized, jurisdictions may simply compete in other areas
more directly under local control and simply over- or underprovide some other public
good or service (Revesz 1997).

20.5.2. Positive Issues and Analysis

While economics provides a theoretically attractive framework for analyzing how
jurisdiction over environmental policymaking ought to be allocated, observed behavior
sharply diverges from its recommendations.

20.4.2.1. Jurisdictional Mismatch

The allocation of authority between the national government and the states in US
environmental law is not tailored to a jurisdiction-externality justification for federal
authority. There are many federal statutes that address environmental problems that do
not have any interjurisdictional externalities. These include federal programs to
remediate hazardous waste sites, and set limits on the allowable level of contaminants
in drinking water. Furthermore, the Clean Air Act and Clean Water Act, which do
address pollution sources with the potential to generate important interstate
externalities, are generally focused on local, rather than interstate, pollution. The core of
the Clean Air Act consists of federally prescribed air-quality standards designed and
implemented at the local level. Indeed, facilities can meet these air-quality standards by
exporting more pollution across state lines (Revesz 1996). Plant-level emissions standards
are not oriented toward facilities with important interstate consequences, instead
covering all sources by pollution category and vintage. The sections of the Clean Air Act
that are specifically targeted toward interstate pollution have been devilishly difficult for
the EPA to implement, with multiple attempts being struck down by the D.C. Circuit Court
of Appeals. The Clean Water Act fares little better, with most of its emphasis placed on
pollution sources that have only intrastate effects (Stewart 1982).

20.4.2.2. Lack of a Global Agreement on Climate Change

If governance power should be allocated to the minimally extensive jurisdiction that
internalizes any relevant externality, in the context of climate change, that jurisdiction is
the entire globe. Because greenhouse gas emissions in any country lead to climate
change risks in all countries, a global approach to greenhouse gas limits is well justified.

There has been a substantial effort through the United Nations Framework Convention
on Climate Change (UNFCCC) to negotiate a set of meaningful mandatory limits on
emissions that would apply to all countries. While the UNFCCC process has had
some successes, it has faced extremely serious stumbling blocks. In particular, the failure
to negotiate a successor agreement to the Kyoto Protocol at the 15th Conference of the
Parties meeting in Copenhagen, Denmark, was seen as a major setback for the Intergovernmental Panel on Climate Change (IPCCC) process (Vidal et al. 2009).

Absent progress at the international level, for a time, initiative for climate policy devolved to the regional or domestic level. The European ETS remains the most robust international emissions-trading program and serves as the primary vehicle for emissions reductions within European countries (European Commission 2013). Domestic efforts to curb greenhouse gas controls have been forwarded successfully in some countries but met with stiff resistance in other countries, such as Australia (Plumer 2013).

### 20.4.2.3. State Innovation

Within the United States, the lack of national emissions limits, and especially the lack of climate legislation, resulted in a further devolution to the state level (Carlarne 2008). US states have adopted three basic approaches. Early efforts tended to involve command-and-control regulation. In 1997, Oregon enacted the first legislation in the United States addressed at limiting greenhouse gas emissions, setting a standard for carbon dioxide emissions from the state’s natural gas electric plants (Environmental Defense Fund 2012). In 2001, Massachusetts enacted carbon dioxide regulations for all power plants, as part of a comprehensive bill aiming to cut pollution from the electricity sector (Daley 2001). In 2002, California enacted legislation that required the state regulatory agency to “adopt regulations that achieve the maximum feasible and cost-effective reduction of greenhouse gas emissions from motor vehicles.”

The California effort was especially significant because the state plays a special role under the Clean Air Act (Carlson 2009). Section 209 of the Act allows California to request a waiver from the EPA to set more stringent mobile source standards than the federal standards. Other states can then follow suit, and depart from the federal standard, if they choose. When California issued regulations in 2004 establishing greenhouse gas limits for motor vehicles, it set in motion a political chain of events that ultimately led to a “car deal” negotiated among major automobile manufacturers, the federal government, and other stakeholders in support of a national greenhouse gas standard for new automobiles under the Clean Air Act (Freeman 2011).

Subsidies to encourage greater reliance on renewable sources of electricity are a second approach adopted by many states. The most common approach, referred to as “renewable portfolio standards,” is a mandate that a target share of the state’s energy supply be generated by renewable sources (Davies 2012). For example, California’s standard requires suppliers to obtain 20% of their energy from renewable sources, with this proportion rising to 33% by 2020 (Farber 2008). Clean energy subsidies face important challenges, including the need to define “renewable” in a manner that accurately captures environmental benefits (Duane 2010) and likely results in relatively high-cost emission reductions (OECD 2013).
Finally, states have experimented with a cap-and-trade approach to greenhouse gas emissions limits. The Regional Greenhouse Gas Initiative (RGGI) is one such effort. RGGI is a collaboration of northeastern states that have signed a joint memorandum of understanding (MOU) setting out each state’s share of a regional carbon dioxide cap, adopted legislation or regulation approving that MOU, and, beginning in 2008, implemented an auctioning and trading process (Duane 2010). While one state from the initial group of ten (New Jersey) has withdrawn, the RGGI regime has remained relatively stable and has raised over $1 billion for participating states in the first four years of operation (Rabe 2009).

In 2006, California passed the California Global Warming Solutions Act, commonly known as AB 32. That legislation launched a multistep process to reduce greenhouse gas emissions in the state to 1990 levels by 2020. The centerpiece of the regulatory approach implementing AB 32 is a statewide cap-and-trade program. California, however, has not adopted a purely market-based approach, having augmented its cap-and-trade approach with a range of additional policies, including plant specific performance standards, energy-efficiency requirements, and a renewable portfolio standard. Many of these additional measures are likely to lead to increased costs without obvious climate benefit (Carlson 2013).

These state efforts set the stage for a renewed round of federal efforts, primarily the Clean Power Plan that was finalized by the EPA in 2015. This rule, promulgated under the agency’s existing authority under the Clean Air Act, establishes state-by-state emissions limitations in part based on prior state experience with emissions reductions. Under the Plan, states have considerable discretion in choosing between different approaches to meeting their emissions budget, a process of experimentation that may ultimately produce information that helps alleviate political gridlock at the national level (Livermore 2017, forthcoming).

The EPA's move to regulate greenhouse gas emissions at the national level in turn helped support efforts by the Obama administration to negotiate a successful emissions reduction agreement during the 21th Conference of the Parties meeting in Paris, France. The Paris Agreement is widely perceived as providing a substantially more meaningful roadmap to genuine greenhouse gas emissions than the products of earlier climate negotiations.

---

20.5. Conclusion

Since the publication of Revesz and Stavins (2007), there have been some significant normative advances in the area of environmental law and economics. For example, the emergence of climate change as the area of central concern for environmental regulation has brought a great deal of attention to the question of how to discount benefits that
accrue into the far future and primarily affect individuals not yet born. Also, the rise of behavioral law and economics has created a shift away from exclusive reliance on neoclassical models.

But the most significant changes have been on the positive side. In particular, the traditional alignment of interest groups has come close to experiencing an about-face. Conservative, antiregulatory groups traditionally favored cost–benefit analysis, market-based instruments, and decentralization. Progressive, proregulatory groups traditionally opposed these approaches. In recent years, however, the tables have often been turned. These shifts suggest that commitment to principles is secondary to commitment to substantive regulatory outcomes, with groups of both sides of the spectrum availing themselves of whatever argument will better promote their preferences concerning the stringency of regulation.

References


Environmental Law and Economics


Environmental Defense Fund, Institute for Policy Integrity, Natural Resources Defense Council, and Union of Concerned Scientists. 2013. “Comment on the Use of Social Cost of
Environmental Law and Economics


Environmental Law and Economics


Notes:


(8) For example, the following bills were proposed during 2011 by the 112th Congress: H.R. 1 (a continuing appropriations resolution for FY2011, which passed the House in February, containing more than twenty riders restricting or prohibiting the use of funds to implement various regulatory activities under EPA’s jurisdiction); H.R. 199, Protect America’s Energy and Manufacturing Jobs Act of 2011 (proposing a 2-year suspension of climate rules); H.R. 457, H.R. 517, H.R. 2018, and S. 272 (to modify EPA’s authority under the Clean Water Act); H.R. 750 and S. 228, Defending America’s Affordable Energy and Jobs Act (pre-empting any regulation to mitigate climate change); H.R. 872, Reducing Regulatory Burdens Act of 2011 (amending the Clean Water Act and FIFRA to alter EPA regulation of pesticide discharge into water); H.R. 910, Energy Tax Prevention Act (to prevent greenhouse gas regulations under the Clean Air Act); H.R. 960 and S. 468, Mining Jobs Protection Act (amending EPA’s consultation procedure under the Clean Water Act); H.R. 1391, Recycling Coal Combustion Residuals Accessibility Act of 2011, and H.R. 1405 (prohibiting coal ash from being regulated under Subtitle C of RCRA); H.R. 2021, Jobs and Energy Permitting Act of 2011 (amending the Clean Air Act to change permitting of off shore sources); H.R. 2250 and S. 1392, EPA Regulatory Relief Act of 2011 (to delay the Boiler MACT rules); H.R. 2584 (an appropriations bill with various riders); H.R. 2681, Cement Sector Regulatory Relief Act (to delay the Cement MACT rules); H.R. 3400 and S. 1720, Jobs Through Growth Act (incorporating several of the above restrictions on EPA authority); and S.J. Res. 27 (a resolution to disapprove EPA’s cross-state air pollution rule).


(12) 40 C.F.R. pt. 98.


Michael A. Livermore
Michael A. Livermore, University of Virginia School of Law

Richard L. Revesz
Richard L. Revesz, New York University School of Law