Transaction Costs and Environmental Policy: An Assessment Framework and Literature Review

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ABSTRACT

This article develops a framework for environmental policy analysis based on an encompassing assessment of transaction costs. This approach emphasizes the *ex ante* costs of establishing environmental entitlements, and the *ex post* costs of administrating, monitoring, and enforcing them. The framework is used to organize a literature review which addresses policy design and instrument choice, as well as optimal environmental policy-making and benefit-cost analysis. The review also considers the empirical literature on transaction costs associated with environmental policy-making, and current practices to record some categories of transaction costs in regulatory impact assessments. The article concludes with a discussion of the implications for environmental policy analysis.

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1 Introduction

This article develops a transaction cost framework for the analysis of environmental policy, and uses it to organize a discussion of recent literature on policy design and evaluation. We also consider the implications of this framework for optimal environmental policy-making and benefit-cost analysis. Our conceptual approach has been influenced by Coase (1937, 1960) and subsequent work in the new institutional economics field by Eggertsson (1990), and others. Some classical contributions in the field of public choice and political economy have also helped to orient the analysis, e.g., Buchanan (1980), Buchanan and Tullock (1975), Buchanan and Vanberg (1988), Maloney and McCormick (1982), Peltzman (1976).

This transaction cost framework will emphasize the costs of establishing the rights to use environmental resources and the costs of the *ex post* monitoring and enforcement of those rights. As such, the focus will extend to all costs related to establishing and implementing environmental policy from the policy's planning and inception through its operational phase. This framework offers ''nothing new'' in the sense that all of the topics within it are addressed in various ways in other literatures; for example, the literature on political economy (Joskow and Schmalensee, 1998; Keohane *et al.*, 1997; Stavins, 2006), the research on monitoring and enforcement (Cohen, 1999; Heyes, 1998), and the instrument choice frameworks which expand the analysis beyond the standard economic efficiency criterion (Fullerton, 2001; Harrington *et al.*, 2004; Goulder and Parry, 2008). What the approach adds is an emphasis on the welfare costs of the political activity surrounding the rights establishment itself — costs which are generally assumed away in the normative environmental policy literature — and a fundamental perspective bringing together distinct aspects of policy-making within a coherent framework. To keep the scope manageable, the article emphasizes the application of this framework to environmental policy-making, rather than to natural resource management, for which the perspective is also relevant.

In the environmental economics field, the term ''transaction costs'' first emerged in the literature on the Coase theorem. In this context, ''transaction costs'' are the costs of bargaining following a rights assignment. Coase also uses the terminology the ''costs of market transactions'' (Coase, 1960, p. 15). This connotation was adopted in the literature which later developed on emissions trading, which assesses the effects of trading transaction costs on the size of markets, the volume of trade, and the overall scope

for market efficiency, e.g., Gangadharan (2000), Solomon (1999), Stavins (1995), Tietenberg (2006). In contrast, the costs associated with establishing and implementing environmental policy, as well as monitoring and enforcing policy *ex post*, have traditionally been distinguished from the term "transaction costs" using such particular labels as "political costs", "administrative costs", "implementation costs", "monitoring and enforcement costs," and the like (Barthold, 1994; Cropper and Oates, 1992; Fullerton, 2001; Harrington *et al.*, 2004). Such costs are also sometimes treated as non-monetized performance constraints, e.g., "administrative burdens" or "political constraints," or their avoidance or minimization viewed as performance goals; for example, "the goal of administrative efficiency," "the capability to monitor and enforce," "the objective of flexible implementation," and so on (Barthold, 1994; Blackman and Harrington, 2000; Fullerton, 2001; Eskland and Jimenez, 1992; Harrington *et al.*, 2004).

Although the traditional association of the term "transaction costs" with the Coase theorem and the literature on emissions trading still seems to predominate in the environmental economics literature, the transaction cost term has started to be employed more broadly to refer to some of the costs just mentioned. In a large survey edited by Freeman and Kolstad (2006), the transaction cost term is used mostly in the narrower sense to denote the transaction costs of emissions trading, but is also used more expansively in one chapter on solid waste management, in which "principal categories" of transaction costs associated with consumer recycling are described as "waste hauling; billing, administrative, and retail systems; consumer costs; and enforcement costs" (Menell, 2006, p. 281). Similarly broad usage is found in Blackman and Mazurek (2001), who refer to the costs of establishing site-specific regulations for EPA's Project XL as "...project development or transactions costs." In evaluating a fee versus standards approach to control automobile emissions, Ando *et al.*, (2007) use the transaction cost label to denote emission inspection costs, and the time and transit costs of vehicle owners to and from inspection stations. But it is in the agricultural economics literature on agro-environmental policy, and in particular on nonpoint source water pollution control, where the expansive usage of the "transaction cost" term now seems most prevalent. A number of papers on the subject use the transaction cost terminology to refer to some combination of the costs of a policy's implementation, administration, monitoring, or enforcement, e.g., Easter (1993), Falconer (2000), Falconer and Whitby (2000), McCann and Easter (1999, 2000); McCann

et al., (2005), Shortle and Horan (2001). The broader connotation of the transaction cost term is also commonly used in the literature assessing climate policy options, e.g., Michaelowa *et al.*, (2003), Repetto (2001) and Woerdman (2001). In summary, the ''transaction cost'' label has started to be applied more expansively, as well as continued to be used in the original, narrower sense to denote the costs of bargaining or market trading.

What accounts for this semantic shift? Two explanations seem plausible. The first is the possible influence of other disciplines on the vocabulary and focus of environmental economists; in particular, the rise of new institutional economics with its central focus on transaction costs as the organizing principle (e.g., Eggertsson, 1990; Williamson, 1981, 2002), and the influence of the law and economics field, whose practitioners are attuned to the institutional and legal issues associated with environmental policy-making (Cole and Grossman, 1999, 2002; Bell, 2003; Richards, 2000). A second possible reason is the growing awareness of policy design and implementation as factors relevant for comparative policy evaluation. Although policy-oriented environmental economists have long recognized the normative import of the way a policy is designed and implemented,[1] a number of recent trends seem to be raising the profile of the issue, including:

- The growing popularity of incentive-based instruments in the United States and in Europe, which has increased awareness of the way design and implementation details affect the performance of this particular class of instruments, and provided a performance record for comparison to prescriptive regulations. (U.S. EPA, 2001; Harrington *et al.*, 2004; Ellerman and Buchner, 2007).
- The state of policy evolution in more-developed countries in which the low-hanging regulatory fruit — the regulation of point source emissions — has largely been picked (and least in so far as local pollution is concerned), leaving more complex regulatory challenges, like nonpoint source pollution, to be addressed. The costs associated with administering, monitoring, and enforcing policies to control nonpoint source pollution are likely to be

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[1] For example, the book "Environmental Improvement through Economic Incentives" (Anderson *et al.*, 1977) systematically assessed possible monitoring difficulties, as well as legal and political obstacles, associated with environmental taxation. (The purpose was to convince the skeptics that environmental charges did not face insurmountable implementation obstacles.) And in their JEL review, Cropper and Oates (1992) encouraged attention to the details of policy design and implementation in order to make theory policy-relevant.

relatively high in comparison with abatement costs, making it natural to focus on the factors which affect such costs.

- The difficult policy design and implementation issues confronting environmental policy makers in less-developed countries, who face relatively weak institutional and legal capacity, financial and political constraints, as well as localized and dispersed nonpoint pollution sources, all of which raise the costs of policy implementation, monitoring, and enforcement (Bell, 2003; Blackman and Harrington, 2000; Eskland and Jimenez, 1992; Russell and Vaughan, 2003).[2]
- The daunting task of developing a policy response to global climate change, the implementation details of which will critically affect the feasibility and cost-effectiveness of global carbon emissions control (Bell, 2006; Michaelowa *et al.*, 2003; Repetto, 2001; Woerdman, 2001).

In summary, the diversification and increasing scope of environmental policy-making has increased the need to apply theory to practice, raising the profile of the costs associated with policy design and implementation, as well as *ex post* monitoring and enforcement. With the broadening of the usage of the term "transaction costs" to apply to these kinds of costs, it is not surprising that the "transaction cost" label, in its more encompassing sense, is beginning to appear more frequently in the literature.

Given this evolution, it seems timely to take stock of the research related to transaction costs and environmental policy evaluation. The extension of the transaction cost construct into the arena of policy design and choice offers a conceptual pay off for thinking about the environmental policy problem, but the cost to date has been the kind of terminological and conceptual inconsistency one would expect in an emerging research area subject to the vagaries of path dependent scholarship influenced by multiple disciplinary approaches. The purpose here is to attempt to provide a conceptual framework which brings out the insights an expanded view makes possible, while hopefully minimizing its associated costs.

To that end, the next section starts by defining the term "transaction costs," and then considers the components of transaction costs, so defined, as

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[2] The level of income and institutional capacity of countries fall along a continuum, raising terminological issues about the labeling of countries' developmental states. The terms "more-developed" and "less-developed" are used in this article to reflect the fact that the level of development matters, without being specific about absolute income thresholds or measures of institutional capacity which would put countries in one category or another.

well as the factors which affect their magnitude. Section 3 turns to the policy evaluation framework which guides the rest of paper. Section 4 shows the significance of environmental policy designs and instruments for transaction costs during the policy establishment period, while Sections 5 and 6 consider the effects of policy designs and instrument choices on transaction costs during the policy's implementation and operational phases. Section 7 shows the relevance of transaction costs for optimal policy-making and benefit-cost analysis. Section 8 turns to the empirical assessment of transaction costs. Section 9 concludes with a discussion of the principal implications, and makes suggestions for future research.

2 Transaction Cost Definition and Components

Coase (1960) first pointed out the relevance of transaction costs for comparative institutional analysis; that is, the extent to which economic activity, including the level of pollution control, could be most efficiently organized through markets, firms, or governments. Other kinds of institutional arrangements, such as informal rules enforced through custom, are now considered in a research agenda on local resource management (Hayes and Ostrom, 2005; Ostrom and Nagendra, 2006). A large literature has also developed on the role of transaction costs in shaping organizational structure (Coase, 1937; Eggertsson, 1990; Williamson, 1981, 2002).

In this article, the tradition of the environmental economics literature is followed in the sense that the government is taken to be the relevant institution, and the role of transaction costs in its organization or efficiency is not considered. In this context, the focus naturally shifts to policy analysis. However, the policy analysis perspective of this article is broader than that in the standard environmental economics literature. The next subsection starts by defining the term "transaction costs" upon which this broader perspective is based.[3] The distribution of transaction costs, so defined, at different stages of policy-making and on different categories of stakeholders

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[3] The terms "transaction cost", "transaction costs", and "transactions costs" appear to be used synonymously in the literature. The term "transactions-cost" is also sometimes encountered, e.g., Falconer (2000).

In this article, the term "transaction costs" is used to refer to the multiple costs associated with a transaction or transactions, while the term "transaction cost" is used to refer to a single cost component. When used adjectivally, however, the term "transaction cost" also has the plural connotation, as in "transaction cost burdens."

is then described. The final subsection overviews factors which affect the magnitude of transaction costs, as they are conceptualized in this article.

2.1 Transaction Cost Definition

Although our focus differs from the emphasis of the new institutional economics, a conception of transaction costs in that literature is borrowed and extended for our purpose. According to Eggertsson (1990, p. 14): "...transaction costs are the costs that arise when individuals exchange ownership rights to economic assets and enforce those rights." The view expressed by Mathews (1986, p. 906) is related: "The fundamental idea of transaction costs is that they consist of the costs of arranging a contract *ex ante* and monitoring and enforcing it *ex post*, as opposed to production costs, which are the costs of executing the contract." These notions seem applicable both to the cost of contracting for the exchange of well-defined rights, as in routine market transactions, and to the costs of creating rights, as occurs in the formation of mergers or other kinds of enterprise reorganizations.

This article extends these fundamental notions to the domain of regulatory policy-making. In this context, the transactions are between "the government," as a representative agent of "society", and the actors subject to regulation. The effect of regulation is to establish use or quasi-ownership rights for some groups of impacted stakeholders.[4] In this context, "transaction costs" are the *ex ante* costs of establishing the use or quasi-ownership rights, and the *ex post* costs of administering, monitoring, and enforcing the new rights arrangement. In contrast, production costs are the costs of carrying out the economic activity the rights establishment incentivizes, i.e., the economic costs of carrying out the regulatory requirements which implement the policy objective.

Turning specifically to environmental regulation, policy-making defines a distribution of environmental use or quasi-ownership rights for polluters and other stakeholders. These rights are given value, either explicitly, by exogenously imposing a price (e.g., an emissions tax), or implicitly, by establishing a regulatory requirement which has the effect of quantitatively restricting

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[4] The term "quasi ownership" is used to reflect the fact that the rights created through regulation — for example, tradable pollution permits — are generally qualified, with the possibility of later regulatory review or modification. The term "licensed property" is sometimes used to refer to this kind of right (Raymond, 2003).

the level of pollution (e.g., a performance standard), creating scarcity rents.[5] An equivalent notion is that environmental policy establishes the level of pollution control, and the degree of compensation or loss for the impacted stakeholders.

Throughout this article, the parallel process of defining a distribution of environmental rights and determining its value is labeled as ''establishing environmental rights'' or ''structuring environmental entitlements'' or simply as ''establishing rights'' or ''structuring entitlements.''[6] Since the rights establishment is effected through various policy designs and instruments, different legal and administrative obligations are associated with particular policy approaches. In this context, then, transaction costs are the *ex ante* costs of establishing environmental policy in all of its aspects, and the *ex post* costs of administering, monitoring and enforcing the policy once established. In contrast, the cost of pollution abatement constitutes the cost of ''executing'' the environmental policy — the ''production cost'' associated with achieving the environmental policy aim.[7]

To provide some perspective, it is worth noting that the source and level of transaction costs in private markets are likely to differ from those arising through regulation. Most importantly, rights negotiations in private markets are typically voluntary, whereas the establishment of rights through regulation typically involves a degree of coercion. Thus, policy-making generates political costs which are absent in private exchanges.[8] The political context is also likely to raise the costs of *ex post* monitoring and enforcement.

Another point is that regulation always establishes new kinds of rights and obligations, since ''failure'' of an existing institution (the ''market'') motivates it. In this sense, the actions of regulators contrast with those of private actors exchanging already-defined rights in well-defined markets. As pointed out before, new rights do emerge in private markets, and it is also true that rights are exchanged through public policy; for example, when assets are bought and sold, or public production is subcontracted. But the balance between creating and exchanging rights is different in the

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[5]   If a performance standard is implemented using tradable permits, the scarcity rent is reflected in the equilibrium price of permits.

[6]   The generic terminology "establishing rights" or "structuring entitlements" could also apply to other regulatory contexts than environmental regulation.

[7]   This definition is similar to the definition in Krutilla (1999). It is also consistent with the definition in McCann *et al.*, (2005).

[8]   Of course, rights disputes do arise in the private sector, and legal costs are incurred to address them.

two contexts — particularly, when comparing regulatory policy-making to private market exchange. Buchanan and Vanberg (1988, p. 102) make the distinction as follows:

> Private responses to externalities, such as the bargaining or the merger solution, operate via some reassignment or rearrangement of rights within a given legal structure or, more generally, within a given framework of socially sanctioned rules and laws. In other words, private corrections are a matter of in period adjustments among market participants — of trades made within a defined institutional context. In contrast to such private responses, political corrections work via some change in the 'rules of the game', they imply some alteration in the rule structure itself.... Politicization, as such, amounts to an abrogation of existing prior legal 'rights' concerning the activity in question. Political corrections are about redefining the rights which the market participants hold, not about trading defined rights....

Since it is presumptively more costly to create or redefine rights than to exchange them, particularly through a political process, the transaction costs of policy-making are likely to be relatively significant.

A third distinction is the different way the value of the rights, and activity levels associated with them, emerge in private markets and the regulatory arena. Using the classic Coasen property rights assignment as the private market analogue, the value of the rights, and associated level of abatement, emerge endogenously through voluntary bargaining. In contrast, policy-making establishes both the value of the rights (level of pollution control) and the distribution of the rights. Thus, at the proximate level, regulation involves the specification of two parameters rather than one. This sharp distinction is ultimately attenuated, however, by the fact that the level of regulation ultimately emerges endogenously through political exchange which stakeholders attempt to influence (see Peltzman, 1976). Specifically in the context of environmental policy-making, polluters can directly negotiate with regulators over the level of pollution abatement to form "voluntary environmental agreements" (Alberini and Segerson, 2002; Glachant, 2005; Segerson and Miceli, 1998) or otherwise attempt to influence the level of regulation. Thus, the way the value of rights emerges in private markets as opposed to government rule-making seems likely to be distinguished principally by the different channels for endogenous actions in the two institutional

contexts, and of course, the different actors involved, and their relative bargaining power, or influence. These contextual differences are likely to have implications for the kinds of transaction costs, and their causes, influencing valuations, with some commonly-discussed sources of transaction costs in the new institutional economics literature, like ''asset specificity,'' less relevant to the formation of ''public contracts'' (laws and regulations) than to the private contracts associated with market activity.

A final distinction applies to a specific class of policy instruments, like emissions taxes or auctioned permits, which create a price for pollution, thereby generating public revenue. Lee (1985, p. 732) makes the distinction as follows:

> The claimants against revenues raised by market prices are generally well-specified, as is the extent of their claims. This is not true with revenue raised by government through political prices. Additional monies raised by government generally go into the common pool of general revenue. The allocation of these revenues among rival interests is determined through competition for political influence, a competition which requires the use of real resources.

In short, the political costs of distributing revenues raised through environmental policy-making can also increase transaction costs.

There are a number of comments and qualifications to make about the conception of policy-related transaction costs described here. First, the term ''cost'' is taken to refer to the standard opportunity cost measure used in welfare economics. By that standard, financial exchanges, such as the payment or receipt of fees, emissions taxes, or fines, or the payment or receipt of subsidies or compensation, do not have efficiency implications *per se.* Of course, resource costs are incurred to raise public funds; these include both political transaction costs associated with alternative ways to generate the revenue, and the commonly-recognized deadweight losses associated with distorting markets (or forgoing other investment opportunities with positive net present values). And as was just pointed out, political transaction costs are incurred to distribute public monies. The transaction costs associated with financial exchanges — in particular, the transaction costs of using environmental policy to raise and distribute revenue — are explicitly considered in the analysis which follows. However, financial exchanges *ipso*

facto sum to zero using the common Kaldor–Hicks criterion, and thus have no efficiency consequence at the proximate level.

A second issue is the relationship between information costs and transaction costs. One view holds that information costs are not the same as transaction costs, but that imperfect information gives rise to costly transactions, so that transaction costs are not likely to arise in a perfect information world, e.g., Eggertsson (1990, p. 15).[9] However, for the purpose of this article, it is not important to distinguish between transaction costs and information costs, either conceptually or semantically. Thus, in the discussion which follows, information costs are commonly labeled as transaction costs. And since information is imperfect and fundamental to many of the transaction costs which arise in environmental policy-making, the transaction cost definition is premised on an economic context in which information is assumed to be imperfect.

Another definitional issue is the categorization of transportation or distribution costs. For example, the transit costs of conveying water moved in response to a water marketing scheme. Some authors categorize such costs as transaction costs (Colby, 1990). However, within the described definitional framework, transportation/distribution costs are one of the costs of executing the rights establishment, not a cost associated with defining the rights themselves, and so do not qualify as transaction costs.[10]

There is an important point to make about the boundary of the transaction cost definition. The fundamental conception upon which it is based logically implies an expansive definition. Consider the definitional implications of a less fundamental conception of rights; for example, the definition of rights at the level of the ownership rights exchanged when a regulated firm makes a required purchase of a continuous emissions monitoring (CEM) device in response to a pollution control regulation. In this context "transaction costs" would include the costs associated with the contractual negotiation about the purchase and delivery of the CEM device, and any

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[9] To make the point that information costs and transaction costs are not the same, the example is used of an isolated individual stranded on an island. This individual would face information costs, but would not face transaction costs.

[10] Thus, "distribution costs" fall under our global definition of "production costs." In this context, "production costs" are construed broadly enough to allow for the possibility of production on the site of the rights definition itself, but would more commonly involve some distribution of inputs to some other site where the production activity itself would actually be conducted.

needed legal actions afterwards to enforce the contract. In contrast, "production costs" would constitute the delivery cost of the equipment and the costs of installing and using it. However, with the concept of rights construed in the fundamental sense noted, all of the mentioned costs would qualify as the transaction costs of environmental policy-making itself; specifically, the cost of measuring compliance with the rules the policy creates. Thus, within the definition implied by a fundamental conception of rights fall all of the transaction costs associated with less-fundamental conceptions, as well as the "production costs" associated with those less-fundamental conceptions.[11]

Although production and transaction costs will be nested across definitional levels, these constructs are mutually exclusive holding the definitional level constant. Taking the fundamental rights definition used in this article, the transaction costs of environmental policy can be defined as the residual category "all costs but abatement costs." Or abatement costs can be defined as the residual category "all costs but transaction costs." The fact that "transaction costs" and "production costs" can range from mutually exclusive to overlapping, depending on the basis for the comparison, is important to keep in mind.

Finally, it is obvious that a residual "not abatement" cost definition for transaction costs is likely to include a broad array of components which might be referred to in a variety of ways, some of them already mentioned in the introduction, e.g., "decision-making costs," " political costs," "legislative costs," "information costs," "implementation costs," "administrative costs," "legal costs," "monitoring costs," "enforcement costs" and like. These labels are used loosely in the literature, with little clarity about their definitions or boundaries (hence little clarity about the degree of mutual exclusivity or overlap they embody). In this article, no attempt is made to clarify such terminological ambiguity, since discriminations at this level are not relevant to the principal emphasis on transaction costs as a broadly-constructed category. But the point should not be lost that all of the terms mentioned embody some component of transactions costs as defined here. In particular, these costs refer to some component of the cost of establishing rights through environmental policy-making, or the costs of the *ex post* administration, monitoring, and enforcement of those rights.

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[11] McCann *et al.*, (2005) offer a similar concept of transaction cost boundaries.

2.2 *Policy Stages and Transaction Cost Incidence*

The fundamental transaction cost perspective implies that all policy-related transaction costs falling on all stakeholders are relevant from the policy's planning and inception through the policy's operational phase. It is useful therefore to more clearly specify the policy stages in which transaction costs occur, and the actors who influence them.

Policy-making might be thought of as a sequence of actions which fall into three stages. The first two effectively disaggregate the *ex ante* period in which rights are established; the third corresponds to the policy's *ex post* operational period.[12]

Stage 1: Policy Formulation and Decision-Making. After a period in which a policy issue is politicized enough to motivate policy action,[13] the policy is planned and defined, and then deliberated and decided. This process imposes a number of costs. Information costs are incurred to assess policy options, to define feasible solutions, and to determine the policy's consequences (McCann *et al.*, 2005). Legislatures and agencies with input into the policy-making incur costs to address stakeholders' concerns and inputs from advocates and experts (Michaelowa *et al.*, 2003). Policies imposing large economic costs on stakeholders, or involving fundamental value conflicts, will induce relatively large resource costs on the part of private actors attempting to shape the decision-making, and the public actors who respond to them. Developing the policy's legal framework can impose significant costs. Overall, ''start up'' transaction costs of all kinds significantly raise the economic costs of environmental policy-making.

Stage 2: Implementation. Once the policy is decided, regulations and guidelines are developed to implement it. Regulations are sometimes specified during the policy formulation itself; for example, the 1992 Clean Air Act amendments established vehicular tail pipe emissions standards (Hahn and Tetlock, 2008). More typically, legislation is crafted to allow flexibility for the development of regulations and discretionary administrative actions during the implementation period. Whether crafted legislatively, or during the

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[12] This stylization follows from Krutilla (1999), and is similar to those used by Thompson (1999) and McCann *et al.*, (2005). Kaplow (1992) also uses a somewhat similar framework. All such frameworks are abstracted from institutional contexts, but are premised on a representative democratic system.

[13] See Lyon and Maxwell (2004, pp. 29–31).

implementation period, the elaboration of the policy's rules imposes a variety of transaction costs, including the information and administrative costs associated with rule-making, and the political costs arising from the actions of impacted stakeholders attempting to influence the policy's definition.

Stage 3: Policy Operation. In the final stage, impacted stakeholders operationalize the policy through their collective responses to the mandates or incentives the policy creates. This is the stage in which abatement costs are incurred, but there are also a number of transaction costs. The policy must be supported through agency administrative actions, and private stakeholders also incur routine costs for record-keeping, accounting, and reporting. The monitoring and enforcement of policy, a collective effort which can involve administrative agencies, the courts, and impacted stakeholders, also imposes significant transaction costs. The economic costs of rent-seeking over revenues raised by pollution taxes or auctioned tradable permits is another potential transaction cost during the policy's operational period. In the United States, agents may offer legal challenges to the policy even during the policy's operational period.

The delineated boundaries between policy-making stages are somewhat arbitrary. The degree to which a policy's rules are elaborated legislatively, as opposed to developed during the implementation period, is itself a policy choice, and some authors construe the policy-establishment period broadly enough to cover actions which are here labeled as falling within the implementation period, e.g., Colby (2000). On the other hand, the term ''policy implementation'' is sometimes construed broadly enough to include the monitoring and enforcement actions necessary to assure the effectiveness of the policy during its operational phase, e.g., Rosenbaum (1980). The precise boundary demarcation between stages (or possible subdivisions within stages) does not matter substantively to the analysis. The focus here is on the cumulative sum of transaction costs over all the policy-making stages, and the stage delineation is only a semantic device to help organize the assessment.

Table 1 displays a stylized pattern of the incidence of transaction costs across policy stages and the principal categories of stakeholders who bear them. The table presumes a particular level of pollution control (or valuation of the rights). The resulting transaction costs are broadly distributed among public entities and private stakeholders, both across and within policy stages. This distribution can be affected by policy-making itself. Mandatory disclosure policies, like the Toxic Release Inventory, reduce the

Table 1. Transaction costs across policy stages.

Policy stage	Transaction cost type and incidence	
	Public sector (national, sub-national, local governments)	Private sector
Stage 1, policy formulation and decision-making	Policy formulation and decision costs for legislative branch Policy formulation and decision costs for agencies	Lobbying costs for polluters, environmentalists, labor unions, consumers, other stakeholders
Stage 2, policy implementation	Cost of regulatory development and legal actions for agencies Cost of legal actions for the judicial branch	Lobbying/legal costs for polluters, environmentalists, labor unions, consumers, other stakeholders
Stage 3, policy operation	Administration, monitoring, enforcement costs for agencies Enforcement costs/other legal costs for judicial branch	Administration, monitoring, enforcement costs for polluters and other impacted stakeholders Legal costs of enforcement and other legal actions for polluters

transaction costs for public agencies associated with collection of environmental information, and reduce information asymmetries, which facilitates collective action by environmental groups (Stephan, 2002). The net impact on transaction costs depends on whether the decrease in public transaction costs is offset by the increase in industry reporting costs and the resource costs of mobilized stakeholders.

Transaction costs can also be shifted from the private to the public sector. Using public resources to help establish electronic exchanges reduces the transaction costs to private actors transacting in markets for tradable pollution allowances, for example. Presumably this shift will reduce the overall level of transaction costs.

Transfer payments can also be used to redistribute the burden of transaction costs among stakeholders, and are an important policy tool for reducing political transaction costs and/or making programs politically acceptable. For example, state beverage container deposit laws in the United States often require distributors to pay collection centers' handling fees. In New Jersey, residents reporting illegal dumping of hazardous waste are granted a fraction of the civil or criminal penalties resulting from subsequent enforcement actions (U.S. EPA 2001).[14]

The general incidence pattern depicted in Table 1 can be illustrated using the particular example of mandated vehicle emissions inspection programs in the United States. All costs related to emissions inspection programs would qualify as the transactions costs associated with monitoring under a broad transaction cost definition.

The federal role in such programs imposes administrative costs on federal agencies, but the states have the responsibility for implementing the programs. Inspections are conducted either in state-owned and operated centralized facilities specialized for emissions testing or in non-specialized privately-owned service stations. In either case, the state will incur administrative costs to monitor the programs, and vehicle owners — the polluters in this case — will bear the time and transit costs associated with the inspection activities. Emission inspection programs have not been popular in the United States, and have been frequently subject to costly legal challenges (Stewart, 2001). In all, vehicle emissions inspection programs impose transaction costs on federal and state agencies, vehicle owners and private or public inspection providers, and the private stakeholders and public agencies involved in or impacted by political activity and/or litigation. This broad-spectrum incidence is not unusual for the distribution of transaction costs associated with environmental policy-making.

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[14] This program was due to expire in January 2011, but has been extended six months. See http://www.nj.gov/oag/newsreleases11/pr20110118-rp1.html.

2.3 Factors Affecting the Magnitude of Transaction Costs

In this section the factors which influence the level of transaction costs are qualitatively described. Define a functional relationship $TC(Q, \mathbf{P}, \mathbf{X})$, where TC denotes the level of transaction costs, Q the level of abatement, $\mathbf{P}$ a vector of policy attributes, and $\mathbf{X}$ a vector of other attributes which influence the level of transaction costs. The $\mathbf{P}$ vector specifically includes parameters for policy design and policy instruments, while the $\mathbf{X}$ vector includes such variables as the state of information, the state of technology, environmental characteristics, economic/institutional contexts, and cultural norms.

Note that TC, Q, and $\mathbf{P}$ are to some extent endogenously determined within a system, along with some components of the $\mathbf{X}$ vector, e.g., the state of information and technology.[15] Given this, the $TC(Q, \mathbf{P}, \mathbf{X})$ relationship is used as an expositional device in several ways. In Sections 4, 5, and 6, $\mathbf{P}$ is viewed largely in an exogenous role; the focus is on the effect of policy-making on transaction costs, holding Q and $\mathbf{X}$ constant. In Section 7, the focus shifts to the relationship between Q and TC, holding $\mathbf{P}$ and $\mathbf{X}$ constant. The context is the way transaction costs affect the optimal level of abatement, and the assessment of net benefits. To provide some perspective for this assessment, the exogenous or partially exogenous factors in the $\mathbf{X}$ vector are now briefly considered, holding $\mathbf{P}$ and Q constant. The variables in the $\mathbf{X}$ vector can shift the level of transaction costs, holding constant the mode and level of environmental policy-making.

2.3.1 State of information

As is well known, imperfect information raises the costs of environmental policy-making in a number of ways. It motivates the assessment of the economic and environmental consequences of environmental policies, including the valuation of environmental damages. It creates the need for *ex post* monitoring to assess polluters' regulatory compliance. Imperfect information is in fact fundamental to the normative distinction between incentive-based

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[15] Consider just the relationship between Q and TC in the context of optimizing the abatement level, and define $MB(Q)$ as the marginal abatement benefits, $MTC(Q)$ as the marginal transaction costs of abatement, and $MC(Q)$ as the marginal abatement costs, with the variables in the $\mathbf{P}$ and $\mathbf{X}$ vectors held constant. The optimum will be determined by the condition: $MB(Q) = MTC(Q) + MC(Q)$, giving MTC and Q, as well as MB and MC.

instruments and prescriptive regulations. With complete information, prescriptive regulations could be tailored to replicate the firm-level techniques and control levels achieved using incentive-based instruments.

Regulatory activity can endogenously influence the state of information e.g., technology-forcing standards incentivize the production of new information. Information costs can also decline over the course of a policy's lifetime, as stakeholders become accustomed to the rules and procedures the policy creates (McCann *et al.*, 2005). This evolution occurs in cap-and-trade programs. Poor information about the value and enforceability of the rights initially discourages polluters from entering the market, but more parties begin trading as ambiguities are clarified with increasing experience (Colby, 2000).

2.3.2 The state of technology

As is well known, the state of technology is a fundamental parameter affecting all categories of costs. In the environmental context, for example, the development of continuous emissions monitoring (CEM) technology in the 1980s made it feasible to mandate CEMs for SO_2 emissions monitoring under EPA's Acid Rain program.[16] CEMs are also required for larger sources in the RECLAIM, and in the recent NO_x Budget Trading Program.[17]

The development of electronic toll collection systems in the 1980s has reduced the costs of charging for roadway use. This technology development has lead to an increase in the use of such systems world-wide (Levinson and Odlyzko, 2008).

The rise of information technology has lowered the costs of policy administration and environmental information exchange. Many environmental databases, such as the Toxic Release Inventory (TRI) maintained by the

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[16] The Acid Rain Program established a cap-and-trade system to reduce SO_2 emissions as part of the Clean Air Act amendments of 1990.

[17] RECLAIM is an acronym for "Regional Clean Air Incentives Market." The RECLAIM began in January 1994. It is a cap-and-trade program to reduce SO_2 and NO_x emissions from utility generators and industrial sources in the south coast basin in the Los Angeles region. The purpose is to help the region come into attainment with National Ambient Air Quality Standards (NAAQS).

The NO_x Budget Trading program was a summertime cap-and-trade system which allowed trading among sources in Northeastern states and Washington D.C to help achieve compliance with NAAQS. The program was superseded in 2008 by the Clean Air Interstate Rule (CAIR), but this evolution has been stalled by legal challenges.

U.S. EPA, and the data on CEM emissions monitoring for the emissions trading programs mentioned above are available online, reducing the costs of public information disclosure.

There is an endogenous component to technology change (a reality which motivates one branch of the field of economic growth theory). In the environmental context, technology-forcing standards provide the most obvious example. Another illustration: the use of CEM technology and the growth of the world-wide web have stimulated software development for record keeping and reporting, generating information both for the public and for the EPA. These developments lowered transaction costs and increased monitoring efficiency (Tietenberg, 2006).

It should be mentioned that technology change, in particular, the expansion of information technology, might also have the effect of raising some kinds of transaction costs. By lowering costs for political actors to organize and mobilize, for example, information technology can increase the transaction costs of the policy's establishment.

2.3.3 Physical/environmental characteristics

Physical and environmental characteristics play a crucial role in the transaction costs of environmental policy-making. A few illustrative examples are considered here. Since NO_x emissions primarily result from combustion conditions rather than from chemical constituents in the fuel, direct emissions monitoring is necessary, or emissions must be estimated from combustion process parameters and fuel flow rates. In contrast, SO_2 or CO_2 emissions are produced from combusting fuel constituents. This difference affects monitoring options and feasible policy instruments. For example, it is possible to impose second-best taxes on the sulfur and carbon content of fuels, avoiding the need for emissions monitoring.

Emissions and effluents from nonpoint sources are more difficult to measure than from point sources, raising the transaction costs of monitoring. Indeed, the transaction costs of monitoring nonpoint source pollution is often prohibitive, requiring the use of second-best policy instruments which are based on pollution proxies e.g., taxes on inputs (Shortle and Horan, 2001). The large number and dispersed nature of nonpoint source polluters also raises the transaction costs of administering pollution controls (McCann and Easter, 1999).

Environmental characteristics affect the spatial and temporal patterns of environmental damages caused by emissions. This has implications for policy design and, in particular, the transaction costs associated with tailoring policy instruments to reflect differentiated environmental effects. Even relatively low transaction cost second-best regulations — for example, tradable permits zonally differentiated by area of environmental impact, or temporally differentiated to reflect peak periods of environmental impact — will be more complex to administer, all else constant, than non-differentiated instruments, which are appropriate in cases where emission impacts are spatially or temporally homogenous. In general, environmental heterogeneities raise the transaction costs of policy-making.

2.3.4 Economic and institutional context

The costs of policy-making are greatly affected by the economic-institutional environment, and a few selective examples are considered here. In the European Union, the transaction costs of implementing policy are increased from different institutional structure and regulatory culture of member countries, and the significant differences in economic status between the older and newer member states, relative to a counterfactual state featuring greater homogeneity. The added jurisdictional layer in countries with federal structures, such as the United States and Germany, affects the level of policy-making transaction costs, raising or lowering them, depending on the relative efficiency of delegating authority within a federal system (see Kerr *et al.*, 1998).

In less-developed countries, formal institutions are relatively underdeveloped, raising the costs of environmental policy-making. Countries may lack the capacity to monitor or estimate point source emissions (Blackman, 2009), and the fraction of the economy in small scale enterprises also makes emission sources harder to track and target (Blackman and Harrington, 2000). Nonpoint source pollution is also a larger problem in less-developed than in more-advanced economies. Beyond the nonpoint source pollution problems encountered in the more-developed world — vehicular emissions, agricultural runoff, and urban storm water runoff — people in less-developed countries sometimes face exposure risks from untreated sewage effluent due to inadequate sewage treatment facilities (Whittington *et al.*, 2008), and

nonpoint source air emissions from residential and commercial heating or cooking sources using biomass-based fuels or coal (the latter a problem in China particularly; see Barnes *et al.*, 2005).

Countries in an early developmental state often raise a relatively large share of their revenue from trade taxes, because it is easier to police and enforce tax collection on goods at the relatively few ports or border crossings through which traded goods flow than to collect taxes from many different firms and individuals (Corden, 1980, pp. 65–66). In countries with weak tax administration, the costs of implementing environmental charges might be higher than the existing institutions can manage (Bell, 2003).

''Lack of political will'' has sometimes been cited as a constraint on the monitoring and enforcement of environmental policy in developing countries (Blackman, 2009). The literature on the environmental Kuznets curve explains this attitude as a natural function of a country's stage of economic development (Aubourg *et al.*, 2008). The environmental Kuznets curve literature implies that less stringent and/or more poorly monitored and enforced policy will be the norm in developing countries falling beneath particular income thresholds.

2.3.5 Cultural norms

Cultural norms also greatly influence the level of transaction costs associated with environmental policy-making. In the United States, the transactions costs of regulatory implementation are increased by procedures for public participation during the rule-making stage, and extensive litigation during and afterwards (Harrington *et al.*, 2004). The rule-making process, as spelled out by the Administrative Procedure Act, requires public notification and agencies generally must consider (but not necessarily act upon) public comments. Litigation can impede the functioning of controversial rules for years, causing agencies to continually reassess and readdress them.

These kinds of transaction costs proximately reflect institutional capacities, but more ultimately, cultural and legal norms supporting citizens' rights. The libertarian strand in American political philosophy provides a natural bias against state-sponsored regulatory activity. The resistance to congestion pricing in many areas of the United States has been attributed in part to the fact that such fees ''conflict with a fundamental and highly

valued belief held by many Americans, namely, that mobility is a right'' (Giuliano, 1992).

Countries having a more consensual approach to policy implementation and/or placing a greater emphasis on collective, rather than individual, interests will have lower political transaction costs associated with establishing environmental policy. In this regard, Japan is perhaps the limiting extreme among more economically-advanced countries. Environmental-policy making in Japan features an extensive network of voluntary agreements (some 33,000 by 1999) worked out between individual firms and local prefectures, as a condition for facility siting. These VEAs differ from those in the U.S. and Europe in two ways. First, they preceded, rather than followed, the development of national legislation. Secondly, they generally contain emission limitations which are stricter than those found in national laws (Welch and Hibiki, 2002).

It is hard to comparatively generalize about the European context, given the diversity of the member states within the EU. However, it seems that Europeans have a less instinctively negative reaction to governmental regulation, and in particular, fees or taxes than in the United States. The normative principle underlying the ''polluter pays'' principle is broadly accepted in Europe (Harrington *et al.*, 2004). The EU has instituted a tradable permit system to control CO_2 emissions — the EU Emissions Trading System (EU ETS) — a major regulatory effort. Begun in 2005, the EU ETS covers large industrial sources and electricity generating units emitting roughly half of the CO_2 emissions produced in the EU (Grubb and Neuhoff, 2006; Ellermen and Buchner, 2007). A similarly-scaled effort in the United States has stalled out in the face of political resistance.

Norms have an endogenous component. Over the past 25 years, incentive-based instruments have moved from the fringe to the mainstream of environmental policy-making (Stavins, 2006). Voluntary environmental agreements, which began emerging in Europe and the United States in the 1990s, have also become increasingly popular. The high cost of the traditional prescriptive approaches, and the growing familiarity and success of incentive-based alternatives, has engendered a significant shift in public opinion. Conceivably, the information exchange from globalization will create an environment for the continuing acceptance of more novel regulatory approaches and a greater convergence of instrument choices across different countries and cultural contexts.

2.3.6 International environmental policy-making

International environmental agreements (IEAs) involve developing a consensus among sovereign states, which increases the transaction costs of policy-making, since sovereign states can neither be forced to join an IEA, nor forced to comply, Heterogeneities in economic, institutional, and cultural factors also raise the transaction costs of collective decision-making required to achieve consensus in international policy-making, and to maintain agreements once ratified. The divide between more-developed and less-developed countries is likely to be especially consequential for global policy-making, given the vast disparities in wealth and institutional capacities around the world. Less developed countries may lack the internal capacity to fulfill international obligations without significant institutional reform (Bell, 2006).

3   Policy Evaluation Framework

For the remainder of this article, the variables discussed in the previous subsection are treated as exogenous, and held constant. On that assumption, an assessment is made of the relationship between environmental policy-making and transaction costs. This section outlines an expanded policy evaluation framework for that purpose. Unless otherwise indicated, the scope is limited to domestic policy-making, or policy-making within the EU.

The first item of business is to review relevant policy evaluation contexts, and to describe their modification within an expanded transaction cost framework. Next, a qualification is offered about the way transaction costs affect the nature of policy evaluation itself. Finally, the described evaluation contexts are related to the structure of the analysis in the remainder of the article.

3.1   Overview of Evaluation Contexts

The task of policy analysis is divided into four evaluation contexts. The first is to assess the design of policy instruments. The second is to evaluate the choice among such instruments, taking into account the way they are designed. The third and fourth evaluation contexts are, respectively, to determine the optimal level of pollution control, and to assess the net benefits of policy alternatives. Each of these evaluation perspectives is discussed further below.

The normative goal of policy design should be to craft policy instruments to minimize the cumulative sum of all policy-related costs across all policy stages for a chosen level of abatement. At the theoretical level, policy stages must be treated simultaneously because policy designs can trade off costs among them. In this broad evaluation, a number of policy design attributes are important. The structure of the environmental entitlement is one fundamental policy design attribute. The degree to which the policy framework is elaborated *ex ante*, or allowed to be developed during the implementation period, is another. Other policy design issues concern the environmental proxy to regulate, e.g., inputs, outputs, emissions, ambient concentrations, environmental damages, etc.; the choice of monitoring technologies; the choice of enforcement mechanisms; and the like. Particular policy instruments will have instrument-specific design issues to address. Emissions trading systems will require the development of trading rules and methods for reporting and recording trading activity. Technology-based standards will require the elaboration of different kinds of pollution control methods.

The second evaluation context is to determine the most cost-effective policy option to achieve a given aggregate level of abatement — the standard instrument choice problem. Traditionally, this evaluation has focused exclusively on abatement costs in Stage 3 of the policy cycle. However, an instrument choice context which includes transaction costs must assess the effects of instrument choices on all costs occurring at all stages of the policy process.

It is important to emphasize that, in a comprehensive evaluation which incorporates transaction costs, choices about instruments cannot be assessed abstractly in the absence of choices about policy designs. This is true at both the theoretical level, and in practical policy-making. When comparing policy instruments at the theoretical level, it is necessary to normalize for fundamental policy design attributes, so that differences among instruments can be cleanly attributed to the properties of the instruments themselves, rather than to particular design features which have not been controlled for. For example, an evaluation of whether incentive-based instruments are more or less costly to implement than more prescriptive regulations is not meaningful without normalizing for the entitlement structure of the alternatives — a key design variable influencing implementation costs. At the level of actual policy-making, decision-makers are likely to be in the position of comparing feasible policy instruments which reflect different kinds of designs. In that case, some combination of the design and the kind of instrument will determine costs and environmental effects. The tasks of policy

design and instrument choice in this context might be undertaken more or less simultaneously, since considering trade-offs among policy designs and instruments could be important for the evaluation.

The theory literature in environmental economics devotes considerable attention to the optimal level of environmental policy-making, and some of the literature on transaction costs assesses the effects of transaction costs on this evaluation (Vatn, 2001). Since some kinds of transaction costs depend on the level of abatement, and transaction costs can also influence the benefits associated with pollution control, transaction costs have the potential to affect the optimal level of pollution.

Benefit-cost analysis constitutes the final evaluation context. Costs of all kinds occurring at all stages of the policy process should be considered in an overall assessment of a policy's net benefits. Since very few benefit-cost studies provide such an encompassing accounting, it is possible that environmental policies are recommended whose net benefits are negative from an expanded transaction cost perspective. This issue is addressed in a later section of this article.

3.2 The Effects of Transaction Costs on the Policy Evaluation

The outlined evaluation contexts implicitly assume that analysts have good information about transaction costs, and operate in a political context in which policy recommendations will influence decision-makers. This conceptual framework provides a theoretically ideal benchmark, but one which must be modified to varying degrees in a positive transaction cost world.

In many cases, the cost of information, as well as other transaction costs, will hinder comprehensive policy analysis, and influence the kinds of normative goals which motivate it. In other words, transaction costs shape the form of the policy analysis itself. For example, trading off compliance goals with the costs of monitoring and enforcement becomes an important analytic task in a positive transaction cost world, since perfect compliance can only be achieved at a cost. And since it is too costly to monitor some environmental performance measures, such as environmental damages, the ''given level of abatement'' in the analysis of policy alternatives must be referenced to a proxy which can be measured — such as emissions — with some tradeoff resulting in the objective to reduce environmental damages.

The optimization of abatement levels may not be feasible in a world of positive transaction costs, since it may be too costly to produce empirical

estimates of marginal costs and marginal environmental improvements associated with increasing pollution control levels. In that case, second-best policy targets will have to be assessed without an explicit optimization.

As well as shaping the form of the policy analysis, transaction costs can affect its utility. In particular, the influence of policy recommendations can be diluted by the political nature of decision-making. Stakeholders are distributed across decision stages, and between the public and private sectors, which diffuses decision-making loci. In a democratic system with public participation, there is no single decision-maker; multiple decision-makers will have there own objective functions and attempt to maximize their perceived self-interests. Policy-making involves a strategic game among self-interested stakeholders, rather than the maximization of a social welfare objective (Welch and Hibiki, 2002).

Policy objectives can be significantly compromised during the implementation process. Administrative impediments are one constraint. There is a chain of decision-makers/choice points from the beginning of the implementation process until the end; lack of coordination, or misunderstanding along this chain can lead to outcomes that depart from the intent of the policy goal, with each step of the implementation magnifying the discrepancy (Pressman and Wildavsky, 1973). A web of principal-agent relationships, in an environment of asymmetric and imperfect information, also has the potential to reshape policy as it moves from the legislative inception to the on-the-ground implementation (Wood and Waterman, 1991).

Symbolism, rather than *bona fide* policy aims, is a possible motive for legislation (Matland, 1995). In that case, policy analysis, as a substantive aid to decision-making, is obviously not very relevant. EU policy-making has been criticized for its symbolic aspect. The common metric for successful implementation has been the transposition of EU directives into the laws of member states (Jordan, 1999). Translating national laws into rules which affect behavior at the local level is often not a priority. A study comparing the national implementation strategies of eight EU member states for the EU directive on ''Integrated Pollution Prevention and Control'' (IPPC) found that compliance was token, essentially maintaining the *status quo* (Bohne, 2008a, 2008b).

Due to the political, bureaucratic, and sometimes symbolic nature of policy-making, the relationship between policy outputs and inputs cannot be taken as a technical production function assumption, as in standard microeconomics. Transaction costs fundamentally affect the form of the

production process itself.[18] This means that the feasibility of policy options, as well as their cost, must be considered in the normative evaluation. But there is likely to be a relationship between transaction costs and the feasibility of policy options. All else constant, the lower the total magnitude of transaction costs of a policy alternative, the greater the likelihood of its feasibility. Thus, minimizing the totality of transaction costs — making trade-offs among particular cost components to minimize the total cost burden — becomes normatively desirable both to increase the likelihood that actions will be feasible, and to reduce the costs of feasible actions. Policy actions which exceed a transaction cost threshold will be rendered infeasible, and thus, will not be part of the class of options for which policy makers can consider in the decision-making.[19]

In sum, at several levels, transaction costs fundamentally affect the objectives, scope, and impact of the policy analysis. But even with the noted qualifications, the evaluation framework described offers some useful perspective. In particular, it invites an expanded focus on transaction costs at all stages of the policy process, and, where feasible, to incorporate information about transaction costs into the policy evaluation. It suggests the possibility of trading off transaction costs falling in different policy stages and on different stakeholders, increasing one kind of transaction costs as an investment in order to reduce another kind, as means to the ends of minimizing transaction costs overall. These themes are pursued in later sections of this article, but for now, three illustrative examples are offered.

Legislation which specifies more contingencies *ex ante* will impose greater transaction costs in the legislative deliberation period, but will reduce at least some kinds of transaction costs during the regulatory implementation. Whether this shift minimizes the cumulative sum of transaction costs across policy stages depends crucially on the nature of the policy problem, and

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[18] This point applies to private enterprises, as well as to the public sector. The literature on transaction costs economics shows that different kinds of firm organizations economize on transaction costs. See Macher and Richman (2008). Since the organization of firms affects input–output relationships, transaction costs are influencing the form of the production process.

[19] As pointed out before, international policy-making is a context where transaction costs significantly constrain the feasibility of action. A main objective of international policy-making is to achieve the maximum participation to prevent the leakage of damaging activities to non-covered areas (Wiener, 1999; Barrett, 2003). Achieving participation at least cost, what Wiener (1999) terms "participation efficiency," thus becomes a particularly important attribute. Low participation efficiency leads to either an inadequate level of participation or achieves an adequate level at a very high cost (Weiner, 1999).

the relative cost of legislative deliberation, versus agency implementation (Kaplow, 1992). This trade off is considered in greater detail in a later section of this article.

The choice of an emissions trading programs based on emission reductions credits, such as the early emissions trading programs instituted under the Clean Air Act in the United States, and the frameworks underling Joint Implementation (JI) and the Clean Development Mechanism (CDM), impose relatively low transaction costs in the policy establishment period, compared to instituting a cap-and-trade program.[20] However, transaction costs at the operational stage of the credit reduction programs are higher than efficiently functioning allowance markets (see Ellerman *et al.*, 2003; Hahn and Hester, 1989). The low political cost of instituting emissions credit reduction programs might explain why such approaches often constitute a precursor stage to cap-and-trade programs — the historical experience in the United States, and conceivably, the future evolution for international CO_2 emissions trading. In sum, the comparison between these two kinds of trading options involves a trade-off between operational transaction costs in Stage 3 and political establishment costs in Stage 1.

As another example, consider a policy proposal to auction a restricted number of environmental rights to achieve an environmental policy goal, versus freely distributing the same number of rights through agency rule-making. Buchanan (1980) argues that political competition in the latter scenario creates the same kind of incentive for dissipating the value of the rights as in the unregulated situation. He suggests that auctioning the rights is the better solution. But unless the auction policy is imposed exogenously, for example, by an executive order, a proposal to auction the rights itself offers a political incentive to dissipate rents — by the actors who would be subject to the auction and would invest resources to lobby against it. As such, a proposal to auction the rights would shift the transaction costs of rent-seeking from the implementation period to the political decision about the policy-making. It will be argued in a later section that this shift is likely to increase the potential for rent dissipation. But yet another alternative would be to grant environmental users an explicit, transferable title at their

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[20] The CDM and JI are both project-based credit trading programs established under the Kyoto protocol. Under JI, an Annex B country (a country committed to CO_2 emission reductions) can earn CO_2 emission reduction units by implementing an abatement project in *another Annex B country*. Under CDM, an Annex B country can earn emission reduction units by implementing an abatement project in *a developing country*, which does not have an abatement commitment.

unregulated use level, i.e., freely distributing an unrestricted level of environmental rights. Then, the agency could purchase the rights and retire them to achieve the desirable regulatory level. In fact, the government of New Zealand used such an approach in the 1980s to initiate a system of individual transferable quotas (ITQs) for fish catch in coastal waters (Colby, 2000).[21] This policy reduced the costs of establishing the program with a degree of political consensus behind it, while imposing some additional implementation costs, since a base environmental use level had to be measured.[22]

3.3 *Applying the Policy Evaluation Framework*

The policy evaluation framework described above is applied in the next four sections of the article. Section 4 assesses the effects of policy design and instrument choices on transaction costs falling in the policy establishment period (Stage 1). Section 5 assesses the effects of policy designs and instrument choices on transaction costs falling in the implementation period (Stage 2). Chapter 6 conducts the same kind of analysis, focusing on the effects of policy-making on both transaction costs and abatement costs occurring in the policy's operational period (Stage 3). Chapter 7 turns to the implications of transaction costs for the optimal level of pollution abatement, and benefit-cost analysis.

It should be emphasized that the choice of policy designs and instruments mostly occur in Stages 1 and 2. It is *the consequences* of those choices which are assessed in the next several sections.

4   The Effects of Policy Design and Instrument Choices on Transaction Costs During the Policy Establishment Period

This section considers the transaction costs associated with the political process of establishing environmental rights. As noted before, transaction costs other than those associated with political behavior are incurred during

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[21] The ITQs were allocated to vessel owners, rather than ship captains or crew, or fishing communities. The process was politically acceptable enough to allow for a relatively low cost program initiation. The program has evolved over time to accommodate changes in fishery stocks, the species regulated, the claims of Maori fisherman, and the like. See Lock and Leslie (2007).

[22] The normative benefit of this outcome requires that taxpayers do not mobilize to oppose subsidizing the industry. The degree to which producer and consumer interests are represented in policy-making is discussed further in Section 4.

the policy establishment period. They include the administrative costs of developing the policy's rules and procedures, and the costs to acquire information about the nature and scope of the environmental policy problem. These kinds of transaction costs are not addressed in this section.

Also noted before are policy design issues relevant for crafting particular policies in addition to those associated explicitly with establishing environmental entitlements. For example, whether the baseline for the regulatory metric is absolute emissions, or relative emissions intensity; whether an emissions trading program includes a banking option; what kind of monitoring system is chosen, etc. Choices about these design attributes will impose their own kinds of transaction costs and also affect the value of environmental entitlements. Such design details are not considered in this section and, as such, contextual detail is lost which might be relevant in particular circumstances. The rational for this quite circumscribed, fundamental focus is to allow conclusions to be broadly drawn and also, to elevate the profile of the welfare costs of the political behavior associated with establishing environmental policy, a fundamental cost which usually gets ignored in the environmental economics literature. The relationship between environmental entitlements and particular policy instruments is also important to clarify, since this relationship sometimes is treated in ambiguous and inconsistent ways.

The first subsection offers some perspective about establishing environmental entitlements, and the way the traditional environmental economics literature has viewed this process. The next subsection assesses the effects of the entitlement establishment on incentives for political transaction costs. The relationship between entitlement distributions and instruments is described. The implications of instrument choice for political transaction costs are then considered. The following subsection compares the relative magnitude of the potential political costs associated with the entitlement structure and those associated with abatement costs. The next section offers some extensions, while the final subsection offers some qualifications.

4.1 Overview

Stakeholders have an incentive to engage in the political process which establishes environmental rights if the rights are valuable enough (pollution control level stringent enough) and organizing costs are low enough. Political disputes around the initial allocation of tradable pollution permits is a

common example in the environmental context (Raymond, 2003). How to initially allocate CO_2 emissions allowances was one of the most contested issues in the negotiation of the EU ETS, and has been a continuing issue in the program's subsequent evolution (Hepburn *et al.*, 2006). The distribution of CO_2 emissions allowances was also a significant political issue in the legislative deliberation in the U.S. House of Representative in 2009 over the Obama administration's proposed cap-and-trade program. In the original proposal, CO_2 emissions allowances were to be auctioned. But in the legislative jockeying which shaped the bill which ultimately passed (the ''American Clean Energy and Security (ACES) Act''), the allocation method was significantly modified, with 75% of the allowances to be freely distributed through 2026. Electricity and local natural gas distribution companies were to receive the bulk of the free allocations.[23]

Environmental economists traditionally have ignored the welfare implications of the political process which establishes environmental entitlements. Instrument choice frameworks have begun adding a ''political feasibility'' criterion as one of several additional qualitative metrics to consider beyond the usual *ex post* efficiency standard (see Fullerton, 2001; Harrington *et al.*, 2004; Goulder and Parry, 2008). But with a few exceptions in the literature on voluntary environmental agreements and self-regulation (e.g., Maxwell *et al.*, 2000; Glachant, 2005), the efficiency cost of distributing environmental entitlements is not explicitly recognized, or addressed in policy analyses. For example, the common recommendation by economists in Europe and the United States to auction CO_2 emissions allowances does not consider the possibility that this policy design could elicit enough economically-costly political opposition to override the suggested *ex post* efficiency benefits.[24]

Why the deflected focus of environmental economists from the welfare costs of establishing environmental rights? There appear to be several reasons. First is the influence of the Coase literature in which the rights assignment is taken as exogenous, and the distribution of rights is not relevant

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[23] This bill passed the House in 2009 but the legislation was never brought to a vote in the Senate. Chances of further action in the near term seem nonexistent, given the current political environment in the United States.

[24] Economists now widely recommend auctioning carbon emissions permits generated through cap-and-trade programs (Hepburn *et al.*, 2006). Particularly influential to this view are the results of general equilibrium modeling showing that environmental regulation increases labor market distortions. See Goulder *et al.*, (1999), Fullerton and Metcalf (2001), Parry and Williams (1999), and Goulder (2002). Cutting labor taxes, while recovering lost revenue from allowance auctions, attenuates the labor market distortion.

for the *ex post* efficiency analysis under the idealized assumptions of perfect information and zero transaction costs. The second welfare theorem is also based on the assumption of exogenous endowments and again, the sharp dichotomy of efficiency and equity obtains under the idealized assumptions. A rationale for dichotomizing the focus on distribution and efficiency is also suggested by the well-known economic principle that using different policy instruments to address different policy aims is most efficient assuming there are no transaction costs to using the different instruments (or transaction costs across the instruments are the same). This logic suggests that environmental policy should focus on *ex post* efficiency, and other instruments should be used independently to address distributional issues. Finally, the comparative statics tradition of standard microeconomics, based on the assumption of costless government intervention and frictionless implementation, naturally shifts the analytic focus to the comparative assessment of states-of-the world before and after the policy's adoption, and away from the intervening period when the policy is established (Krutilla, 1999).

In a decision-making context featuring endogenous policy-making and positive transaction costs, however, it is not logical to exclude policy-making costs from the normative assessment. Specifically in the environmental policy context, it is not conceptually consistent to justify policy on the welfare consequence of the self-interested actions of actors *ex ante* in the absence of legally-established rights, and to evaluate the policy *ex post* on the basis of the welfare consequences of actor's self-interested responses to the incentives the policy creates, while ignoring the normative consequences of self-interested political actions of the very same group of actors in the intervening period when the policy is established (Krutilla, 1999). The fact that the economic stakes of allocating valuable property can be significantly higher than the *ex post* efficiency differences commonly thought to be normatively significant in the conventional environmental policy evaluation makes this point empirically, as well as theoretically, relevant.

The following subsection turns to the way the entitlement is structured and its relationship to policy-making to bring out the normative significance of this important step in the policy-making process.

4.2   *Structuring the Environmental Entitlement*

Structuring an environmental entitlement has the potential to impose significant financial liabilities, or gains, on stakeholders, and to incentivize costly

pollution abatement. These effects constitute the gross pay-offs to devoting economic resources to attempting to influence the policy-making. Expected payoffs, rather than gross payoffs, will have the closest link to the actual resource costs of lobbying, since there is a probabilistic element to political outcomes. Other factors will influence the relationship between gross payoffs and political costs; a qualification is offered at the end of Section 4. For now, gross payoffs are taken as a positive index of rent-seeking costs, as in the traditional rent-seeking literature.

The focus is also on the incentive effects the entitlement structure offers to polluting firms to lobby over the policy-making.[25] Political economy models frequently emphasize the influence of firms, rather than tax payers or consumers, due to the fact that financial liabilities and costs are concentrated on firms, and the relatively large number of consumers or taxpayers creates organizational barriers to political mobilization (Buchanan and Tullock, 1975; Maloney and McCormick, 1982; Peltzman, 1976). The economic effect of environmental regulation on producers has also traditionally featured prominently in the equity norms of legislators (Raymond, 2003). The crucial implication of this assumption is that displacing the policy's impact from polluting firms to other stakeholders — for example, by granting polluters a relatively large share of the environmental entitlement — will reduce the net resource cost of political activity. This assumption is qualified at the end of this section for the particular case of allocating CO_2 emissions allowances.

What about environmentalists as a factor in the political activity around environmental policy-making? Environmentalists historically have focused on one component of the entitlement structure — the level of pollution control — with less attention to the other — the distribution of the environmental rights. That stance has put them in the position sometimes of supporting policies, like conventional performance standards, which reduce pollution while compensating polluters with significant share of the environmental entitlement (Maloney and McCormick, 1982). This possibility is taken up later in this section.

For now, the effect of the structure of the environmental entitlement on polluters is considered, before turning to firms' abatement costs. Adapting Farrow (1995, 1999), Krutilla (1999), and Pezzey (1992, 2003), the financial

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[25] It is assumed that polluters are firms, abstracting from the reality that consumption behavior can also generate pollution.

effect of the entitlement can be generically expressed as:

$$\theta \equiv P(e - \alpha \mathrm{eo}) = Pe - (\alpha P)\mathrm{eo} = Pe - (\alpha \mathrm{eo})P \qquad (1)$$

where "θ" is the proximate financial impact (indicating a loss/gain by a positive/negative number); "P" is the pollution price; "e" is the level of emissions post environmental policy, "eo" is the polluter's baseline emissions level before environmental policy, and "α" is a parameter which determines the polluter's share of the emissions entitlement ($\alpha \in [0,1]$).[26] The equilibrium condition $P(e) = \mathrm{MB}(e)$ is assumed to underlie Equation (1), with $\mathrm{MB}(e)$ being the marginal net benefit of emissions (the value of an emissions right). It is assumed that $P(0) = \mathrm{MB}(0) > 0$, $\frac{\partial P}{\partial e} = \frac{\partial MB}{\partial e} < 0$, and $P(\mathrm{eo}) = \mathrm{MB}(\mathrm{eo}) = 0$.

The economic context determines the degree to which θ is an absolute loss relative to the polluter's financial position before regulation, or instead, the foregone opportunity to earn supernormal returns. If the demand for the polluter's product is infinitely elastic, the burden of the policy falls fully on the polluter. In that case, $\mathrm{MB}(e)$ can be interpreted as the marginal value of emissions in production, and pollution control costs are equal to lost profits on the units of pollution reduced. The other limiting extreme is that the firm's costs are constant, so that all the burden of the environmental policy is passed on to consumers.[27] In this case, $\mathrm{MB}(e)$ can be interpreted as the marginal value of emissions in consumption, and pollution control costs as equal to the lost consumer surplus on the units of reduced pollution. Of course, the typical case would involve some degree of burden sharing between polluters and consumers.[28] For now, we abstract from the implications of the degree of burden-sharing between firms and consumers in terms of the impact of gross rent-seeking payoffs, following the traditional assumption of the rent-seeking literature that firms have the same incentive

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[26] Note that the baseline is for actual emissions, rather than an emissions rate (quantity of emissions per unit of some input or output). A rate-based baseline is not uncommon in environmental policy; for example, performance standards in the United States to reduce air emissions use this kind of baseline, as does the "Refunded Emissions Payment" (REP) program in Sweden to control NO_x emissions. The distinction can be important in practice (see Kuika and Mulderb, 2004; Laurikka, 2002; Sterner and Isaksson, 2006; Fischer and Fox, 2009). But the essential insights are illustrated here using the absolute emissions baseline.

[27] See Fullerton (2001) for a model using this interpretation.

[28] It is entirely the market configuration which determines the burden-sharing between polluters and consumers, not how entitlements are distributed. The incentive effect in Equation (1) is the same whatever the degree of the firm's entitlement share under our assumptions.

to lobby to capture supernormal returns as to avoid absolute financial losses (Tullock, 1988). This assumption is qualified subsequently. Additionally, the usual assumptions needed for a partial equilibrium focus are also initially adopted.[29] The conclusions derived from this approach do not change significantly when empirical results from the general equilibrium literature are subsequently considered.

In Equation (1), the level of regulation, e, or equivalently, the pollution price $P(e)$, and α are the two policy parameters. We focus first on the α parameter, which determines the entitlement distribution. Then we consider the P parameter which gives the value of the entitlements.

4.2.1 Distributing the entitlement

The α parameter can be thought of in several ways. One way is to define α as the share of the firm's baseline emissions received as an emissions entitlement post regulation. That assumption is easily seen in the left-most representation in Equation (1), and is convenient to adopt here.[30] With $(\partial \theta / \partial \alpha) = -Peo < 0$, the firm's financial liability is inversely related to its share of baseline emissions received as an entitlement, holding all else constant.

Abstracting for the moment from transaction costs or institutional constraints which would restrict choices, the emissions entitlement αeo could theoretically be implemented using a number of different alternatives (see Table 2).[31] To illustrate, first consider the case where $\alpha > 0$, so the firm receives an emissions entitlement which is some positive share of baseline emissions, but that the entitlement granted, αeo, is less than the firm's desired emissions level post regulation, i.e., Case 1 in Table 2, where $e - \alpha eo > 0$. This entitlement might be implemented as an emissions standard at αeo, with a certain unit fine of P imposed on the excess $e - \alpha eo$

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[29] That is, that whatever the elasticity of demand, the demand curve does not shift with policy; that the regulated sector is too small in the overall economy for regulatory adjustments in the sector to change the prices of inputs, and that lump sum taxes are available for offsetting fiscal impacts.

[30] Alpha could also be defined as the fractional rate of the pollution payment (P) to provide unit compensation (αP) for each of eo emissions, or to provide the basis for a lump sum rebate to the polluter (conveniently seen from the middle term in Equation (1)). For purposes of practical policy-making, the implementation of this concept might be useful (Farrow, 1995).

[31] Table 2 adapts Table 1 in Krutilla (1999).

Table 2. Entitlement structure of different policy instruments.

Entitlement fraction/implicit entitlement	Proximate financial impact on polluter	Corresponding policy instruments and their entitlement
$(\alpha = 0)$; $(\alpha eo = 0)$	Pe	Zero emission standard with unit fine of P on all emissions
		Conventional emissions tax of P
		Emissions tax-subsidy, with P as the emissions tax, and no lump sum rebate
		Tradable permits, all bought by polluter at price P in an auction
		Coasean property assignment of eo to pollution-impacted party
$(0 < \alpha < 1)$; $(\alpha eo < eo)$	Case 1 $e > \alpha eo \rightarrow$ $P(e - \alpha eo) > 0$	Emissions standard at αeo with unit fine of P on $e - \alpha$ eo additional missions
		Emissions tax, with αeo infra-marginal emissions exempted; tax of P on $e - \alpha eo$ emissions
		Emissions tax-subsidy, with P as the emissions tax, and lump sum rebate of αeoP
		Tradable permits, with αeo permits grandfathered; $e - \alpha eo$ permits purchased by polluter
		Coasean property rights sharing with αeo to polluters and $(1 - \alpha)eo$ to pollution damaged parties
	Case 2 $e = \alpha eo \rightarrow$ $P(e - \alpha eo) = 0$	Conventional emissions standard
		Emissions tax with αeo infra-marginal emissions exempted
		Emissions tax-subsidy, with P as the emissions tax, and lump sum rebate of αeoP
		Tradable permits, with αeo permits grandfathered, no permits purchased/sold
		Coasean property rights sharing with αeo to polluters and $(1 - \alpha)eo$ to pollution damaged parties

Table 2. (*Continued*)

Entitlement fraction/implicit entitlement	Proximate financial impact on polluter	Corresponding policy instruments and their entitlement
	Case 3 $e < \alpha eo \rightarrow$ $P(e - \alpha eo) < 0$	Emissions standard at αeo, with over compliance unit subsidy on $(\alpha eo - e)$ excess emissions Emissions tax-subsidy, with P as the emissions tax, and lump sum rebate of αeoP Tradable permits, with αeo permits grandfathered, and $(\alpha eo - e)$ sold by polluter Cosean property rights sharing with αeo to polluters and $((1 - \alpha)eo$ to pollution damaged parties
$(\alpha = 1)$; $(\alpha eo = eo)$	$P(e\text{-}eo) < 0$, for $e < eo$	Emissions standard at eo, with over compliance unit subsidy on $(eo - e)$ excess emissions Conventional Pigouvian emissions reduction subsidy of P on $(eo - e)$ emissions Emissions tax-subsidy, with P as the emissions tax, and lump sum rebate of eoP Tradable permits, with eo permits grandfathered, and $(eo - e)$ permits sold by polluter Cosean property rights assignment of eo to polluter

Terminology:
e are polluter's emissions post regulation.
eo is polluter's baseline emissions before regulation.
P is pollution price — the of level emissions taxes/subsidies, unit fines/subsidies, or the price of tradable permits.
α is fraction of baseline emissions to which the polluter is entitled (post regulation), $\alpha \in [0, 1]$.

emissions.[32] Alternatively, the entitlement αeo could be implemented as part of a tradable permit policy which grandfathers αeo emission entitlements. In this case, the firm would buy $e - \alpha$eo emissions allowances at price P to cover the excess emissions above the entitlement.[33] Third, an emissions tax could be imposed which exempts αeo infra-marginal emissions, while imposing an emissions tax of P on the $e - \alpha$eo excess emissions (effectively the equivalent of the first alternative). Fourth a tax-subsidy scheme could be implemented, which imposes an emissions tax of P on all emissions, for a financial liability of Pe, while making a lump-sum rebate back to the firm of the amount αeo$P$. For the sake of completeness, it is worth pointing out that each of these alternatives has the same effect as a Coasean assignment of αeo emissions entitlement to the polluter under the idealized assumptions of the Coase Theorem. The other cases in Table 2, indicating different levels of entitlement, can be interpreted analogously.

This kind of theoretical symmetry among instruments, and in particular, between price and quantity-based instruments has been noted in Farrow (1995, 1999) and Pezzey (1992, 2003).[34] With a marginal incentive effect given by a common pollution price P, the entitlement distribution for any policy can be defined without affecting the policy's short-run efficiency characteristics. Moreover, any of the alternatives displayed in Table 2 will have the same long run efficiency effect and impact on industry size, provided that αeo is granted in perpetuity as a *bona fide* property right which, like capital or other fixed factors, the firm could sell upon exiting the industry (see Farrow, 1995, 1999; Pezzey, 1992, 2003). In short, all of the policy options in Table 2 are formally equivalent in terms of their *ex post* efficiency effect, under the assumed first-best conditions.[35]

Given that all of the policy alternatives in Table 2 are normalized with respect to their Stage 3 efficiency effects, a clean comparison can be made

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[32] This type of fine — effectively a pollution tax above a threshold — is not common in practice. More common is to set the expected value of infrequent sanctions high enough incentivize compliance with the standard. This issue is discussed in Section 6.

[33] The symmetry between tradable permits and the other options ignores the possibility that permits could be banked, opening the door to inter-temporal trading. The implications of extending the analysis inter-temporally are not considered here.

[34] This discussion has been conducted at the firm level, but the efficiency properties hold in the industry provided that quantity controls are complemented with an exogenously-imposed pollution price, or implemented through tradable permits.

[35] This conclusion holds in a first-best setting, and in a world of perfect certainty. As is well known, second best-settings, and uncertainty over marginal abatement benefits and costs, can introduce policy-relevant distinctions between quantity and price-based instruments (Weitzman, 1974).

of their differing Stage 1 political establishment costs, assuming again a positive relationship between the polluter's financial liabilities and lobbying costs. Independently of the instrument which implements the entitlement, political costs will decrease from the top row of the table, where the polluter receives no entitlement, to the bottom row where the polluter is granted a full environmental use entitlement. There are several implications. First, the conventional normative instruments ranking based on Stage 3 efficiency effects is biased in the omission of an important variable — the way the entitlement is structured during the policy's establishment period. This design variable must be explicitly considered when assessing the welfare cost of any policy instrument, and should be normalized in any comparative assessment among policy options (Krutilla, 1999). Or, in other words, the welfare costs of environmental policy instruments cannot be evaluated or ranked independently of the way the policies are designed. Statements like ''incentive-based instruments are more or less politically feasible or more or less easy to implement than prescriptive regulations'' are not meaningful unless the comparative assessment has been normalized along other relevant dimensions — the entitlement structure in this case.

Secondly, the link between the economic costs of the policy and its entitlement structure provides another channel through which the distributional aspect of policy-making cannot be divorced from the policy's overall efficiency effect. Again returning to the example, policies are becoming less economically costly moving from the top to the bottom of Table 2, holding the Stage 3 efficiency effect constant, because, under the stated assumptions, progressively more of the environmental entitlement is being distributed to the polluter.

To the extent that policy choices are not constrained, the theoretical symmetry among instruments illustrated in Table 2 gives policy-makers the flexibility to choose entitlement distributions to reduce the political transaction costs of emissions taxes, as well as tradable permits, or to choose selectively between price-based and quantity-based instruments, if one of these options should happen to be preferable in a particular context for other reasons (Farrow, 1995, 1999; Pezzey, 1992, 2003).

The implementation of a tax-subsidy scheme in Sweden known as the ''Refunded Emissions Payment'' (REP) program provides an example in practice. This policy is designed to reduce the emissions intensity of NO_X production from energy plants and other industrial point source emitters (Sterner and Isaksson, 2006). Each firm pays an emissions tax and, after

deducting 0.2–0.3% of the revenues to cover the program's administrative costs, all of the residual revenue is refunded back to the group of firms who paid it in proportion to each firm's output level (in this case, the firm's useful energy consumption).[36] This rebating method yields a net subsidy to firms who are able to lower their emissions intensities below the industry average, while imposing a net tax liability on those whose emissions intensities are greater than average. This effect diffuses political resistance since the efficient part of the industry gains from the policy, while enabling a tax rate which is high enough to discourage NO_X pollution.[37]

As of 2006, the charge amounted to $6000 per ton of NO_X emitted, compared to permit prices in the U.S. NO_X Budget Trading Program in existence at that time which amounted to several hundreds of dollars (Sterner and Isaksson, 2006). This charge level is high enough to qualify the emissions tax in the REP program as a *bona fide* incentive-based instrument. As such, it departs from most environmental charges throughout the world, which are typically regarded as user chargers, with rates too low to affect behavior (Barthold, 1994; U.S. EPA, 2001; Harrington *et al.*, 2004). The Swedish program is a promising example of the way in which policy design can be used to minimize both abatement costs in Stage 3 and policy establishment costs in Stage 1.

4.2.2 The value of entitlements (Level of regulation)

The level of regulation obviously affects abatement costs, but in perhaps less obvious ways, also affects the financial impact of the entitlement on polluters. To see this, differentiate (1) with respect to the pollution price P and simplify to yield:

$$\frac{\partial \theta}{\partial P} = e(1 + \varepsilon) - \alpha e o = (e - \alpha e o) + e\varepsilon \qquad (2)$$

where ε is the price elasticity of demand for emissions along MB, $P(e) = MB(e)$, $\varepsilon \equiv (\partial e/\partial p)(P/e)$, with $\varepsilon \leq 0$. Equation (2) is the sum of two

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[36] The incentive effect is somewhat different than rebating revenue based on emissions, but becomes more like an emissions-based rebate system the larger the industry and the smaller is any firm's share of total output. For the details, see Sterner and Isaksson (2006).

[37] As for all policies based on emissions rates rather than absolute emissions baselines, aggregate emissions can increase under the REP program. However, the output base has the advantage of reducing the impact of the policy on competiveness, and minimizing negative second-best market effects (Sterner and Isaksson, 2006; Fischer and Fox, 2009).

terms: the common term for marginal revenue, $e(1 + \varepsilon)$, and the polluter's entitlement offset, $-\alpha$eo. If the polluter has no emissions entitlement, $\alpha = 0$, $-\alpha$eo in Equation (2) drops out. In that case, increasing the level of regulation from eo ($P = 0$) will impose a marginal financial loss on polluters up to the point where the polluter's demand elasticity for emissions is -1. For more stringent regulation, the pecuniary effect of the entitlement distribution will actually decline (though of course abatement costs will continue to increase). At the other extreme, if the polluter is entitled to its original emissions level, $\alpha = 1$, then $(\partial\theta/\partial P)$ is unambiguously negative, e.g., $((e - \text{eo}) + e\varepsilon) < 0$. As regulation becomes more stringent, the value of emissions rights increase $((\partial P/\partial e) = (\partial MB/\partial e) < 0)$. Fully compensating the polluter for all lost emissions at the value of emissions when they are scarce offers a net financial gain. Intermediate degrees of emissions entitlement will give a more mixed picture.

This result has policy implications. Specifically, negotiating with polluters for more stringent pollution control in exchange for a greater share of the emissions entitlement will allow for more stringent pollution control objectives to be met while minimizing the firm's financial liabilities, and degree of political resistance, reducing political transaction costs. This behavior appears to occur endogenously in many cases. Environmental policies have historically distributed a relatively large share of the entitlement to polluters.

4.3   The Choice of Policy Instruments

The focus to this point has been on the pecuniary component of environmental policy. However, the policy's abatement costs also offer a gross incentive to contest the policy *ex ante*, and it is well known that different policy instruments, as conventionally defined, impose different abatement costs. Yet, as formulated in Table 2, all policies have the same *ex post* abatement effect. Thus, if policy instruments could be normalized in this way, there would be no difference in the incentives of different instruments to induce polluters to contest environmental policy. In that case, the entitlement distribution would be the sole variable affecting political transaction costs, as was discussed in the previous section.

The necessary and sufficient conditions for a regulator to be able to exercise the range of policy choices implicit in Equation 1 and shown in Table 2 is to be able to set the α and P parameters independently, and to either

establish P exogenously (as through a pollution tax or emissions reduction subsidy) or to enable P to emerge endogenously through a permit trading program. For the remainder of this article, it will be assumed that this flexibility is constrained or not fully exploited. Thus, we will revert to describing policy instruments in the conventional way. In this lexicon, "prescriptive regulations" are technology-based mandates or conventionally-defined performance standards targeting emissions.[38] And as is standard, the term "incentive-based instrument" will be used to denote emissions taxes or subsidies, tradable permits or credits for emissions reductions, or the other kinds of well-known instruments like deposit refund system, liability rules, and the like. Defined as such, it is well known that incentive-based instruments will give the most flexibility for *ex post* economic adjustment among firms during the policy's operational period. The economic benefit of this flexibility should reduce political resistance during the policy establishment period, and lower policy-making transaction costs *holding the entitlement distribution and level of pollution control constant, i.e., holding the structure of the environmental entitlement constant.* Thus, for example, one would expect a conventional emissions standard to impose higher political transaction costs than a policy which implemented the same initial standard by granting the firm an equivalent number of tradable pollution permits. This is on the assumption that the market for tradable permits functions well enough for firms to be able to use it.

This reality opens the door to bargaining with polluters about policy instruments and compliance levels and schedules. In fact, all tradable permit programs to date have featured some kind of bargaining in which firms have accepted the implementation of stringent regulations or accelerated compliance deadlines in exchange for the flexibility to trade emissions allowances. This regulatory context is when cap-and-trade programs are most politically feasible (Colby, 2000). One example is the trading program in lead rights instituted by the EPA in the 1980s to phase out lead from the gasoline pool (hereafter denoted as "EPA's lead trading program"), for which compliance deadlines were accelerated. Another is the development of a market for tradable SO_2 allowances under EPA's Acid Rain Program to implement stringent schedule of SO_2 reductions (Tietenberg, 2000). The greater flexibility of tradable permits has also been used by regulators to bargain for

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[38] If emissions standards are stringent enough, of course, they can force the choice of particular technologies.

more stringent pollution reduction targets in the RECLAIM Program. The recent NO_x Budget Trading program was designed to help bring nonattainment regions in states of the northeast, and Washington D.C., into compliance. In summary, the choice of incentive-based instruments, which minimize abatement costs, as well as the choice of policy designs which structure entitlements to minimize polluter's financial liabilities, can be used to reduce transaction costs in the policy's establishment period and to help achieve environmental goals.

4.4 The Impact of the Entitlement Structure versus Abatement Costs

Abatement costs are typically the focus of the conventional analysis. But the sum of the financial liabilities associated with the entitlement structure and the polluter's abatement costs provide the gross rent-seeking incentive. It is of general interest to see how the relative magnitude of these components compare.

Consider the financial liabilities of an environmental policy in the limiting cases when the polluter is granted no emissions entitlement, $\alpha = 0$, giving the financial liability (Pe). This liability in fact could be substantially larger than the polluter's abatement costs because the price for all infra-marginal emissions is P, giving Pe as the policy's total financial effect, while the equilibrium condition $P = \text{MB}(e)$ and the assumption of rising marginal abatement costs means that marginal abatement costs will be less than P on all infra-marginal units of abatement. In short, the proximate financial impact on the firm of entitlements with the structure of emissions taxes or auctioned permits must be larger than the magnitude of abatement costs unless the percentage of emissions reduced through abatement is relatively high — more than 50%. Thus, it is only for very stringent regulatory programs requiring large, non-marginal changes in abatement — like EPA's lead trading program, or the phase-out of CFCs in response to the Montreal Protocol — where the firm's abatement costs could be larger than the financial effects of policies with the entitlement structure of emissions taxes or auctioned tradable permits.

Market adjustments in a general equilibrium setting do not change this conclusion. Economy-wide pecuniary effects have the potential to dwarf the net efficiency effects defined in conventional welfare measures. Consider the results of a general equilibrium analysis of a policy to cut carbon emissions by about 23% over the 2002–2080 period in which all of the allowances

are auctioned to upstream firms, with the revenue used to cut labor taxes. This policy generates a present value for the allowance revenue of about $3.54 trillion, while imposing economic efficiency costs of about $1.48 trillion (Goulder, 2002). Thus, the value of allowances is about 2.39 greater than the policy's welfare cost. For the same emissions control scenario, grandfathering the permits to firms reduces revenue to zero while raising the efficiency costs to about $2.81 trillion. The efficiency cost difference between the policy alternatives is thus about $1.33 trillion. This difference is often used to justify the recommendation to auction the permits and use the revenue to offset revenue losses from cutting distortionary taxes. But the relative difference in the pecuniary component firms experience is about 2.66 times larger than the efficiency difference (3.54/1.33). This suggests that political transaction costs the auction policy could potentially incentivize could be of a magnitude to be policy-relevant.

4.5   Discussion and Extensions

The assessment of this section has emphasized the potential political costs of structuring environmental entitlements. But since there is a direct relationship between the entitlement structure and the revenue the policy raises, the topic could have been viewed equally as about the potential transaction costs of structuring environmental policy to raise revenue. Following the well-known efficiency principle cited earlier — that using multiple instruments to achieve multiple aims is likely to be normatively best, holding transaction costs constant — the evaluation of policy proposals to structure environmental entitlements to raise revenue should also consider alternative ways of producing this revenue. Perhaps fractionally increasing some tax in the economy would raise the same revenue at lower cost than auctioning CO_2 emissions allowances, for example. Crucial to this comparison would be to assess both the commonly-recognized economic distortions — the normative basis for proposals to auction CO_2 emissions allowances — and the universally disregarded political transaction costs. The sum of these two components provides the relevant normative content for the comparative analysis of alternative ways to generate revenue.

   The premise of this section has been that granting some entitlements to polluters will reduce their incentive to rent-seek around the policy's establishment. However, freely distributing allowances creates its own rent-seeking incentives (Nordhaus, 2007). This point is consistent with the Buchanan

view discussed earlier that bureaucratic mechanisms for distributing limited rights will dissipate rents. However, while some rents will be dissipated when government agencies freely distribute limited rights, as polluters lobby to increase their share of the free distribution, the potential costs will likely be lower than those associated with policy designs which grant polluters none of the entitlement, as when all of the rights are auctioned. Zero entitlement auctions impose non-marginal losses of significant magnitude, while the difference in the expected number of freely-allocated entitlements typically imposes comparatively marginal losses. Since the latter provides less of an incentive to rent-seek than the former, it is not reasonable to point to the rents dissipated through freely distributing entitlements as a justification for auctioning all of them.

This point can be demonstrated using Equation (1). Suppose that α_1 is the minimum entitlement share a firm can expect to receive if the entitlements are freely distributed, and α_2 is the maximum, with $\alpha_2 > \alpha_1 > 0$.[39] As such, $(\alpha_2 - \alpha_1)\text{eo}P$ is the potential financial liability (stated as a positive number) of receiving the lesser entitlement share α_1 rather than the greater share α_2.[40] On the other hand, the financial liability associated with receiving no entitlement share is Pe. This will always be greater than $(\alpha_2 - \alpha_1)\text{eo}Pe$, provided $\alpha_2 - \alpha_1 < e/\text{eo}$. Since e/eo is the emissions share on which polluters incur liabilities when all the rights are auctioned ($e/\text{eo} * \text{eo} = e$), the condition shows that rent-seeking incentives will be less for free distributions whenever the potential range for the distribution is less than this amount. That condition seems likely to hold in most cases.[41]

We now turn to the special case of regulating CO_2 emissions, which differs from the standard assumptions made in this section up until now. First, the financial effects of regulating upstream CO_2 emitters can be significantly

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[39] Such a range can be specified explicitly, rather than surmised on the part of the prospective recipient. In the policy discussed before which established ITQs in the New Zealand coastal fishery, minimum and maximum possible quotas were computed before the program's initiation. This allowed vessel owners to anticipate their possible quota range once the policy was implemented. See Lock and Leslie (2007, pp. 13–14).

[40] Define θo as the polluter's financial liability with no entitlement granted, and θ_1 and θ_2 as the financial liabilities corresponding to freely granted entitlements based respectively on entitlement shares of α_1 and α_2, with $\alpha_2 > \alpha_1 > 0$. Using Equation (1), $\theta o = Pe$, $\theta_1 = Pe - \alpha_1 \text{eo}P$, and $\theta_2 = Pe - \alpha_2 \text{eo}P$. Stated as a positive number, the additional financial liability of receiving a freely distributed entitlement corresponding to α_1 rather than α_2 is: $\theta_1 - \theta_2 = (\alpha_2 - \alpha_1)\text{eo}P$.

[41] Since $e/\text{eo} < 1$, the required condition would fail when $a_2 = 1$ and $a_1 = 0$. But it does not seem likely that the expected range for freely distributing entitlements would extend all the way from zero to the firm's historical emissions level.

passed on to downstream producers and consumers, so the compensation required to cover the losses of upstream firms turns out to be a relatively small fraction of the revenue which would be raised through auctioning all CO_2 emissions allowances (Hepburn *et al.*, 2006; Goulder, 2002; Grubb and Neuhoff, 2006; Sijm *et al.*, 2006). A common recommendation to auction some share of CO_2 emissions allowances presumes that upstream firms will accept just the share of supernormal returns needed to compensate for the costs of reducing carbon emissions, forging the opportunity for further rent-seeking. This assumption seems plausible, but bears further study.

The magnitude of scarcity rents generated from policies to control CO_2 emissions are much larger than for traditional regulatory programs, and the price effects in energy markets are visible to downstream industrial energy users, final consumers, and regulators. Thus, downstream industrial energy users and other consumers likely have more of an economic incentive than typically assumed in political economy models to politically engage around the entitlement structure. And legislatures are likely to be relatively receptive to these interests. Recent efforts to control CO_2 emissions in the United States seem to reflect these realities. As mentioned earlier, the ACES Act distributes emissions allowances freely, but the bulk of the allowances go to downstream industries rather than upstream producers. The recently-initiated Regional Green House Gas Initiative (RGGI) provides for quarterly allowance auctions, with the proceeds used to finance energy efficiency and renewable energy investments. [42] It also seems increasingly likely that CO_2 emissions allowances will be auctioned in the future in the EU ETS (Ellerman and Joskow, 2008). These developments suggest a change in the way CO_2 allowances might be allocated compared with traditional allocation methods to control local pollution.

Of course, the fact that different allocation methods for CO_2 emissions allowances are becoming more politically feasible does not *ipso facto* make them normatively desirable. The standard welfare metric must be used to make that judgment. The high visibility of energy price effects, as well as the large and broadly-distributed economic effects of CO_2 regulation, will increase the welfare cost of distributing CO_2 allowances by any method

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[42] The RGGI is a state-sponsored CO_2 emissions cap-and-trade program covering 10 northeastern and mid-Atlantic states. The goal is to reduce CO_2 emissions by 10% from the power sector by 2018. The ninth auction was held in September 8, 2010. See: www.rggi.org/home.

(U.S. CBO, 2003). This reality justifies explicit economic analysis of the political costs of alternative allowance allocation alternatives.

4.6 *Qualifications and Conclusions*

Formal modeling of the political economy of environmental policy-making is needed to provide a better picture of the possible outcomes and normative consequences of different policy designs and instruments. In addition to gross rent-seeking payoffs, the probabilistic outcomes of political competition, institutional/legal features of the political environment, the costs of organizing political activity, and information asymmetries are important variables to consider.[43] The discussion so far has also presupposed that agents commit real resources, rather than campaign contributions or bribes, to attempt to influence political outcomes. The latter, of course, are financial transfers which do not impose resource costs. This issue complicates the evaluation of the actual transaction costs of establishing environmental policy.

An expanded modeling context will show that polluters sometimes have incentives to agree to voluntary environmental agreements, rather than to engage in a political contest; they also might have an incentive to preempt policy-making through self-regulation. The increasing use of voluntary environmental agreements, and industry-developed performance standards, such as ISO 14000, shows this behavior in practice. These approaches have the potential to lower the transaction costs of structuring environmental entitlements, though possibly at the cost of less-stringent pollution control. The incentives for less-conflictual policy approaches are similar to those of disputants in a legal conflict, where one party faces a lawsuit, or the risk of a lawsuit, and thus faces the choices to compromise in advance, preempting the law suit, or settling the case out of court, or continuing the dispute in a legal proceeding (see Cooter and Rubinfeld, 1989). Applying this framework to environmental policy-making, the decision would involve a two-stage game, where polluters first self-regulate or attempt to negotiate a voluntary environmental agreement and, failing that, attempt to influence the ensuing political contest. It is worth pointing out the main issue here is whether

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[43] See Keohane *et al.*, (1997) for an analysis of the different influences on environmental policy-making, including the particular attributes of the political-institutional context which affect policy outcomes. Maloney and McCormick (1982) and Buchanan and Tullock (1975) are earlier works on the political economy of environmental policy.

agents implicitly or explicitly negotiate over the entitlement structure, or instead, contest it. In a world of endogenous policy-making and positive transaction costs, these decisions are more relevant than bargaining over the entitlements *ex post*, as in the standard Coase literature (Jung *et al.*, 1995).

Glachant (2005) develops a bargaining model which reflects the features described. In this model, the equilibrium solution is a voluntary agreement; indeed, the polluter accepts a higher level of regulation than would have occurred through explicit regulation. What makes this outcome economically rational is the polluter's cost-savings when the political contest is avoided. However, this model does not consider the transaction costs of negotiating or implementing the voluntary agreement. Settlement costs are a key variable determining whether agents settle legal disputes (Cooter and Rubinfeld, 1989); the analogue transaction costs for voluntary environmental agreements are also likely to be important.

Segerson and Micelli (1998) develop a model where polluters face a probabilistic threat of environmental regulation, and the possibility also of subsidies to encourage voluntary abatement. The level of pollution control emerging in equilibrium voluntary agreements depends on the value of such parameters as the relative bargaining power between agents and the social cost of the subsidies.

In a model of firm self-regulation, it is found that firms can profitably self-regulate if the level of abatement they credibly commit to is sufficient to preempt the consumer groups from lobbying for more stringent formal regulation (Maxwell *et al.*, 2000).[44] Preempting the political contest avoids the resources firms would otherwise incur to lobby against the regulation. In this model, the welfare costs of self-regulation, when it emerges as an equilibrium strategy, are lower than when regulation emerges legislatively.

In general, it seems likely than when firms self-regulate or negotiate voluntary agreements, transaction costs of structuring environmental entitlements will be reduced, though possibly at the trade off of the level of regulation. But just as only a fraction of legal disputes are preempted or settled *ex ante*, the scope for settling environmental policy conflicts through bargaining or self-regulation is likely to be circumscribed. Further research is needed to

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[44] This model is based on a three-stage game rather than a two-stage game. In Stage 3, the firms establish their output level.

provide greater clarity about the scope for non-conflictual environmental policy-making, and its relative benefits and costs.

5 The Effects of Policy Design and Instrument Choices on Transaction Costs During the Implementation Period

In this section, the focus turns to the effects of policy design and instrument choice on transaction costs imposed during the policy implementation period. Different aspects of the implementation process are studied in a number of different fields, including public administration, law and economics, and environmental policy. This section reflects insights from relevant research both inside and outside the environmental economics literature. Again the focus is somewhat selective, emphasizing the fundamental aspects of policy design and instrument choice, rather than the less fundamental details of particular policy approaches.

5.1 Policy Design

Policy design influences transaction costs during the implementation period in a number of ways. A fundamental issue is the degree to which a policy's content is specified *ex ante* through more prescriptive legislation, or is less prescriptively drawn so that the policy's content emerges *ex post* through the actions of regulatory agencies or the courts, and/or the decentralized behavior of locally-impacted stakeholders. This issue is considered in a number of different literatures.

The public administration literature features a debate between a "top down" school and the "bottoms up" school (see Matland, 1995). The former subscribes to the notion that legislatures should enact reasonably specific policy objectives, and that implementation constraints are normatively undesirable barriers to legislative aims. The "bottom up" group believes that policy-making should arise from the local level and, in fact, that local factors are often crucial in shaping policy outcomes. The implementation flexibility of EU directives seems consistent with the "bottoms up" philosophy, e.g., the decentralization of watershed management planning in the context of the EU Water Framework Directive (Kastens and Newig, 2007).

The normative distinctions in the discussion of "top down" and "bottoms up" policy-making approaches parallel issues raised in the

separate literature on the appropriate jurisdictional level for environmental policy-making (e.g., Oates, 2001; De Oliveira, 2002; Kerr *et al.*, 1998). The institutional literature on the efficacy of local institutional arrangements for the management of local resources (Hayes and Ostrom, 2005; Ostrom, and Nagendra, 2006) seems consistent with the ''bottoms up'' perspective, as does the literature on collaborative and adaptive management (e.g., Ruhl, 2006).

Another perspective emerges from the law and economics literature. A fundamental study by Kaplow (1992) describes conditions under which the more ''top down'' or ''bottoms up'' mode of policy-making is economically rational in terms of the way legislation is structured. Kaplow shows that when the policy problem applies to frequently occurring actions presenting similar ''fact conditions'' — for example, the routine disposal of dry cleaning fluids — the transaction costs of legislating general rules which will apply ''wholesale'' to all cases *ex ante* are lower than the transaction costs of determining the appropriate procedures more flexibly *ex post*, through regulatory rule-making, or litigation. Moreover, under these conditions, the information costs to individuals of learning the requirements of the legislative rules are relatively low, yielding some combination of better compliance, or lower compliance costs.

On the other hand, when the regulated actions are heterogeneous and infrequent, it is costly to determine all relevant fact contingencies *ex ante*. In this case, drafting legislation flexibly enough to allow regulatory agencies and the courts to shape the implementation of the policy *ex post* is normatively desirable.While this kind of policy design will increase transaction costs during the implementation period, this shift reduces the sum of transaction costs over the policy's establishment and implementation.

Another perspective derives from political economy. From this perspective, the implementation period offers self-interested stakeholders another opportunity to attempt to shape the policy through political or legal actions. An example is wetland protection under the Clean Water Act, which covers ''navigable waters'' and ''waters of the United States.'' [45] The Army Corp of Engineers' regulatory interpretation about what specific types of wetlands are included under these terms has been the subject of intermittent litigation since at least 1985 (US v. Riverside Bayview Homes, Inc).[46] Since

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[45] Clean Water Act, §502(7), 33 USC §1362(7).
[46] United States v. Riverside Bayview Homes, Inc. 474 U.S. 121 (1985).

it first issued regulations defining ''waters of the United States'' in 1975, the Corp has revised its rules concerning wetland coverage several times. Most recently, in a 2006 decision (Rapanos v. U.S.),[47] the Supreme Court overturned the Corps' criteria for wetland coverage requiring a continuous surface connection with a permanent navigable water. Agency staff had to again redraft rules and procedures based on this new interpretation.

More flexible implementation provides incentives for rent-seeking, shifting some of the transaction costs which would have been incurred to contest the policy to the implementation period itself. However, inflexibly-imposed rules can also be legally challenged in the implementation period, effectively continuing the political dispute from Stage 1. The effect of more or less flexibly drawn regulations on the sum total of rent-seeking across policy stages seems ambiguous.

Policies which are politically acceptable during the establishment period often face relatively low political transaction costs during the implementation period. In this case, the entitlement structure can be viewed as an important design variable for facilitating the implementation of policy goals, just as it is for minimizing political resistance during the policy's establishment. However, the relationship between the acceptability of policy during the establishment and implementation periods is complicated by number of factors. Legislative requirements are sometimes kept deliberately ambiguous to aid the passage of controversial legislation, effectively shifting political conflicts to the implementation period (Matland, 1995). Unless this trade-off is pushed to the limiting extreme, both the policy's establishment and its implementation are likely to face political challenges. Structuring entitlements to reduce political costs in this case is likely to reduce transaction costs at both stages of the policy-making.

It is also possible that policies which are politically acceptable at the establishment phase become controversial during the implementation as stakeholders learn more about them. Additionally, some kinds of stakeholders, like environmentalists, may have little influence at the national level but have more bargaining power during the implementation period; thus, the implementation period is the stage chosen to influence the policy-making. In sum, the relationship between the political acceptability of policies in Stage 1 and the subsequent implementation period is not straight forward.

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[47] Rapanos v. United States, 547 U.S. 715 (2006).

Turning to other policy design issues, it seems intuitive that more stringent policy goals would be more difficult and costly to implement than less stringent goals, and this claim is sometimes made (Rosenbaum, 1980). But this conclusion only holds if not enough entitlements are assigned to the polluter when the policy is designed. As discussed in Section 4, more stringent regulation increases the value of entitlements; hence, polluters gain from greater regulation if they are awarded enough environmental entitlements to compensate for losses. Thus, conclusions about the effect of a policy's stringency on implementation transaction costs cannot be made without considering the way environmental entitlements are distributed.

Of course, the administrative complexity of policies influences transactions costs. In this context policy-specific design details particularly matter. As an example, an empirical study of policy alternatives to control area source agricultural run-off in Minnesota found that implementing a management system with a technical assistance component, such as contour farming, would impose higher transaction costs through agency-to-farmer contact than developing infrastructure, such as a waste containment system. (McCann and Easter, 2000). For programs depending on voluntary participation, such as agro-environmental programs which subsidize farmers to implement environmental controls, minimizing the transactions costs which fall on private parties is a necessary condition to attract participants into the program; hence, a necessary condition for implementation success (Blackman and Mazurek, 2001; Falconer, 2000).

The policy baseline also affects transaction costs, just as it affects abatement costs.[48] In addition to ''second best'' issues related to pre-existing distortions in markets impacted by the policy, the degree of change from the policy *status quo* affects transaction costs.[49] Extensions of existing programs generally impose lower administrative costs than initiating completely new programs, by sparing start up costs and taking advantage of existing agency expertise (McCann and Easter, 1999). The status quo may also have psychological value through the ''endowment effect'' (Jacques, 1992), and

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[48] For example, the existence of pre-existing distortions in the economy raise the abatement costs of any environmental policy option (Goulder *et al.*, 1999).

[49] The discussion here reflects a distinction between a change from the policy *status quo*, versus a change from the abatement *status quo* for a particular policy type. Increasing the abatement level for any particular policy, such as increasing the stringency of an emissions standard, will increase abatement costs, and possibly transactions costs. On the other hand, making a change in the kind of policy, holding the abatement level constant, will only increase transaction costs.

accepted equity norms (Raymond, 2003). The evolution of incentive-based instruments and voluntary environmental programs in the U.S. and Europe following the establishment of regulatory programs has likely economized on transaction costs relative to starting such programs in the absence of a preexisting regulatory foundation (Hahn and Hester, 1989).

5.2 *Instrument Choice*

This subsection considers the implications of instrument choice for transaction costs falling in the implementation period. The fundamental policy design details are held constant in the comparative assessment; in particular, the entitlement structure of policy alternatives and the degree of *ex ante* versus *ex post* specification. It is also assumed that the degree of entitlement sharing between polluters and impacted parties is such that firms face their abatement cost without full compensation. This assumption can be taken as a default for many environmental policies observed in practice.

Under these assumptions, there are several fundamental, inter-related channels through which the transaction costs of implementing incentive-based instruments are likely to be lower than for conventional regulatory alternatives. First, for incentive-based instruments, decisions about the choice of pollution control techniques and firm-specific pollution control levels are endogenously made by polluters in the policy's operational stage. In contrast, for conventional regulatory alternatives, these decisions are made to varying degrees by regulators during the policy's implementation period. In a world of perfect or near perfect information and low or no transaction costs, regulators would be able to elaborate and administer the specific standards to replicate, or nearly replicate, the results of endogenous polluter responses to incentive-based instruments. In that case, shifting the locus of decision-making from polluters in the policy's operational stage to regulators in the implementation period would have little normative consequence. But precisely because information is imperfect and transaction costs are positive, the regulatory costs of elaborating the market outcome achieved by incentive-based instruments are likely to be prohibitive. This means that feasible regulations will not be differentiated significantly to reflect different pollution control costs in the regulated industry. As is well known, this regulatory structure will increase abatement costs during the policy's operational period. But even relatively undifferentiated standards are also likely to impose substantial decision-making, information, and administrative costs

during the implementation period, since regulators assume responsibility for decisions about firm-specific pollution control levels, techniques, or methods. Of course, these relative costs will in general be higher for technology-based mandates than for performance standards.

This reasoning suggests that, as a default, the sum of abatement costs in Stage 3 and implementation transaction costs in Stage 2 are likely to be lower for incentive-based instruments than conventional regulatory alternatives, holding all else constant. This view is consistent with the assumption that market-based mechanisms generally accomplish the resource allocation task more efficiently, and at lower administrative and informational cost, than centrally-directed administrative rationing through governmental rule-making.

As in the policy establishment period, political resistance to the implementation of incentive-based instruments is likely to be comparatively low — again under the crucial assumption that that the entitlement structure is normalized across instruments. Polluting firms confronted with high-cost prescriptive regulations during the implementation period have no way to avoid these costs other than to contest the regulations which impose them. In contrast, incentive-based instruments allow for adjustments to be made through economic mechanisms in Stage 3 (Ellerman, 2006). Firms with high abatement costs can buy additional pollution allowances, or can pollute more and pay an emissions tax. On the other hand, prescriptive regulations only provide an incentive for firms to petition for waivers (Ellerman, 2006), an action which imposes transaction costs. Incentive-based regulations also give polluters in individual cases an incentive to over-comply, by selling or banking pollution permits, for example. Firms have no incentive to go beyond compliance with conventional regulations (Ellerman, 2006).

The implementation of environmental policy in the United States exemplifies these points. Prescriptive regulations under the Clean Water Act and the Clean Air Acts have been subjected to a raft of high cost legal challenges, slowing implementation (Harrington *et al.*, 2004). Regulatory gridlock is one reason behind the rise of incentive-based instruments, particularly, cap-and-trade programs. As mentioned before, cap-and-trade programs arose as a way to accomplish stringent regulatory objectives, or to accelerate compliance schedules. Regulators were willing to adopt the incentive-based approach out of recognition of the high administrative and political costs of the regulatory alternatives required to achieve environmental policy goals

(Ellerman, 2006).[50] With the credible threat of the less-preferred mode of regulation in the background, polluters also had the incentive to accept the more preferred incentive-based means to the accomplishment of the policy goal (Colby, 2000).

The default assumption that incentive-based instruments are more implementable than more prescriptive approaches might not always hold. To provide some perspective, here are three possible counterexamples. First, if polluters are not better informed about pollution control costs than regulators, incentive-based instruments could lose their relative advantages. For example, private automobile users might not have the technical knowledge or information to understand the cost-effectiveness of different control options to reduce vehicular emissions. In that case, emission fees are not necessarily more cost-effective than standards (Ando *et al.*, 2007). And it is not clear that they would be easier to implement. To generalize from this example, emissions arising through the consumption activities of dispersed and uniformed consumers might not yield the normative benefits typically assumed for incentive-based instruments, including the advantages of relatively low-cost implementation. On the other hand, such a context would also offer incentives for the market entry of knowledgeable and specialized brokers. Brokers would add additional transaction costs, but would improve the overall cost-effectiveness and implementation feasibility of incentive-based approaches in this context.

The experience of SO_2 emissions regulation in Germany provides another counterexample. Stringent control measures based on technology-based standards were successfully implemented to reduce acid deposition in German forests. Firms complied with these standards and even negotiated more stringent reductions ''voluntarily.'' In this case, overwhelming political support reduced the bargaining power of firms, and technology-based standards were the only feasible option to achieve stringent control targets. In these particular circumstances, the more prescriptive approach is likely to have been as cost-effective and implementable as an incentive-based alternative (Watzold, 2004).

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[50] For example, there were 130 specific control measures required in the 1989 Air Quality Management Plan for the LA basin which polluters otherwise would have had to comply with had the RECLAIM not have been instituted (Ellerman, 2006).

Policy-specific attributes will also matter to the comparative assessment. In this regard, the fixed costs of starting the market institutions which underlie cap-and-trade program can be significant (Colby, 2000). No doubt, the costs of designing markets for pollution allowances can be too high in some contexts for this policy approach to be implementable.

Notwithstanding these specific examples, the logic that incentive-based instruments will be more implementable than more prescriptive approaches, holding policy design constant, is likely to hold true in many cases.

It is also worth considering the implementation costs of voluntary environmental agreements, taken as a distinct class of policy instruments. It was previously noted that the negotiation of voluntary agreements can lower political transaction costs in Stage 1, possibly at the cost of relaxing the environmental policy goal (e.g., Alberini and Segerson, 2002). However, the theory literature on voluntary agreements has not captured the potential for relatively high administrative costs during the implementation period. The negotiation and implementation of voluntary agreements can require a significant resource commitment from regulators and participating polluters. The history of EPA's Project XL project illustrates the limiting extreme. Started in 1995, Project XL was billed as an innovative policy approach, allowing polluters to customize regulations to their own particular circumstances, provided the agreed-upon limits went beyond compliance. However, the program bogged down from the high transaction costs associated with EPA approval of site-specific regulations, and general bureaucratic inefficiency (Blackman and Mazurek, 2001). Due to lack of participation and high transaction costs, the program was terminated in 2002.[51]

This experience suggests the possibility that voluntary environmental agreements do not necessarily have the implementation advantages of the standard incentive-based instruments. If voluntary programs take the form of Projext XL, in fact, the implementation challenge is greater than that of conventional performance standards. Of course, voluntary agreements constitute a diverse category, so some kinds of voluntary agreements might be relatively easy to implement. This is an issue which deserves additional research.

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[51] Cultural factors are also relevant to the assessment, however. As mentioned before, voluntary agreements negotiated between local prefectures and firms are routinely negotiated in Japan (Welch and Hibiki, 2002).

6 The Effects of Policy Design and Instrument Choices on Transaction Costs During the Policy's Operational Period

This section explores the effects of policy designs and instrument choices on transaction costs in the policy's operational stage. The emphasis is on monitoring and enforcement costs. When monitoring or enforcement are not feasible, the focus shifts to the possible trade-offs among various kinds of costs and objectives, e.g., transaction costs, abatement costs, and environmental policy goals. Also considered are the transaction costs of using revenue raised by environmental policy. The transaction costs of using different policy instruments are also assessed, as well as the transaction costs associated with emissions trading, a topic which has attracted considerable attention in the literature.

Administrative costs are given relatively little emphasis in this analysis. In the policy's operational period, administrative costs are typically a combination of costs related to particular policy design attributes, or fixed costs associated with policy-making in general. In either case, it is difficult to say much at the theoretical level. However, in some case it is possible to make generalizations about administrative costs based on the number of actors regulated through policy-making.

Conceivably, administrative costs could be relatively insignificant compared to monitoring and enforcement costs. The cost of using CEMs for monitoring SO_2 emissions under the Acid Rain Program in the United States are relatively high compared to administrative costs (Keohane, 2009). However, administrative costs are empirically significant enough to be the focus of policies in the United States and the EU to reduce them. That issue is taken up in Section 8.

6.1 Policy Design

This subsection takes up three issues. First, an assessment of the design of monitoring and enforcement systems is made, in terms of the implications for transaction costs. Then, second-best policy designs are assessed when emissions monitoring is costly or infeasible. Lastly, the transaction costs of using environmental revenues are considered.

6.1.1 Monitoring and enforcement

The functioning of environmental policy requires monitoring the compliance of polluters with the rules the policy establishes and imposing sanctions

when violations occur. Monitoring actions impose direct costs, and also administrative costs associated with record-keeping and reporting. Polluters can also incur costs to evade detection of their non-compliance. Enforcement involves all necessary actions to sanction violators and bring regulated parties back into compliance. Associated costs include the resources expended to impose and contest sanctions, and monitoring and follow-up to incentivize future compliance or the remediation of environmental damages.

A substantial literature addresses optimal monitoring and enforcement strategies (see Cohen, 1999 and Heyes, 1998). Two kinds of policy objectives are considered. One is to determine optimal penalties to maximize social welfare. The other is to determine optimal penalties to minimize monitoring costs to induce compliance with a given standard, or to improve compliance rates. For either of these contexts, the analysis is usually premised on the standard theory of rational choice under uncertainty, first applied in the enforcement context by Becker (1968). Some research considers expanded frameworks which include compliance motivations beyond those of the standard rational-choice model, considering, for example, the effects of social norms (May, 2005).

Applying the standard rational-choice perspective to the simplest case, optimal unit penalties are shown to satisfy $X = \pi P$, where X is the marginal value of the action the monitoring is designed to deter, π is the probability that a violation is detected, and P is the unit penalty imposed when violations are discovered. Risk-neutral polluters will not pursue an action if its marginal value is less than the expected marginal penalty πP, or, equivalently, if the marginal penalty is equal to or greater than $P = X/\pi$. If the goal is to incentivize firms to internalize the marginal environmental damages (MD) associated with violations, for example, then the penalty can be set at $P = MD/\pi$.[52] If the goal is to deter a firm from exceeding an emissions standard, then the penalty can be set at $P = MC/\pi$, where MC is the firm's marginal abatement cost (the marginal benefit to the polluter from non-compliance) at the emissions standard.[53]

This simple framework gets extended in many directions. Research commonly assesses the effects of shifting some of the monitoring responsibility from regulators to firms, for example, through self-reporting require-

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[52] Heyes (1998) refers to this as the "Pigouvian" penalty.

[53] If monitoring and sanctioning is perfect, then $\pi \to 1$, and the two penalty structures described revert respectively to a Pigouvian tax and an emissions tax set at the rate of the firm's marginal abatement.

ments backed by penalties for false reporting (Livernois and McKenna, 1999; Stranlund *et al.*, 2002); monitoring polluters differentially based on their compliance history (Harrington, 1988); expanding the sanctioning options from financial to non-pecuniary penalties, such as criminal sanctions (Heyes, 1998); and monitoring some aspect of a polluter's compliance efforts other than emissions rates (Cohen, 1999 and Malik, 2007).

Regardless of the features of particular models, several important implications emerge. First, the goal of complete compliance is not generally economically efficient. It is worth some sacrifice of the environmental goal when the marginal benefits of stricter compliance are less than marginal monitoring and enforcement costs.[54] Second, the degree of this trade-off can be influenced by policy design. The original Becker article showed that the penalty level could be used as the variable to increase deterrence, economizing on costly monitoring actions. However, legal and political constraints restrict the degree to which penalty levels can be increased, so the strategy to trade off the level of the penalty for monitoring costs has practical limits.[55] Another strategy for economizing on monitoring costs is to partially shift pollution monitoring from regulators to firms, as mentioned above, by requiring firms to self-report emissions. This policy design will reduce the regulator's monitoring cost while increasing the polluter's, and also add the costs of preparing and auditing the polluter's emissions reports. But overall monitoring costs are often lower, and monitoring efficiency higher (Stranlund *et al.*, 2002).

Another important implication from the monitoring and enforcement literature is that deterrence incentives operate at the margin. Thus, the distribution of entitlements, a crucial variable affecting transaction costs in Stages 1 and 2 of the policy cycle, should not affect the enforceability of policies during their operational period. But the price of pollution, or the level of regulation, does matter, since the marginal gain to polluters from going beyond compliance is the economically relevant variable for specifying a sanctioning regime.

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[54] This trade off can be carried too far, or not far enough. Various political economy models suggest less-than-optimal enforcement activity, although some yield excess enforcement (Cohen, 1999). Additionally, if monitoring is imperfect "giving false positives" for noncompliance, polluters may be induced to over-control pollution risks (Heyes, 1998).

[55] Wealth constraints and risk aversion of firms may also restrict the level of penalty which can be feasibly applied (Cohen, 1999).

6.1.2 Transaction cost trade-offs with imperfect monitoring or enforcement

Devising monitoring systems can be quite complex in practice. First-best would be to monitor environmental damages directly. This goal would have to be based on firm-specific standards for each polluter's emissions, or polluter-specific pollution taxes or permits, differentiated by the environmental damages of the emissions.[56] Since the administrative costs of tailoring regulatory systems to this extent are prohibitive, and the associated monitoring would not be feasible, a blunter differentiation of regulatory effort is practiced.[57] One approach is to base regulations on emissions and differentiate regulations by regional areas or sub-regional zones depending on environmental conditions. This approach reduces the transaction costs of the regulatory design and allows compliance to be judged through emissions monitoring. The RECLAIM program illustrates this approach. The trading area is divided into a coastal zone and an inland zone. Polluters in the coastal zone are only allowed to buy permits from within the zone, whereas polluters in the inland zone can by permits from either zone (Gangadharan, 2000).

Unless emissions are of the uniformly-mixed type, treating emissions as a proxy for environmental damages will impose costs relative to the theoretical benchmark of an ideally differentiated system. These costs are in the form of lower environmental performance, higher abatement costs, or some combination.

In many cases direct emissions monitoring itself is not feasible. Indeed, direct emissions monitoring is relatively rare. Continuous emissions monitoring systems (CEM) are the state of the art and, as mentioned earlier, are beginning to be used more frequently — particularly in emissions trading programs for local pollutants. As might be expected, CEMs are costly, though costs are declining with technological advances. The high capital costs of CEMs now render them feasible only for large industrial point sources, or utility generators.

In the absence of continuous emissions monitoring, the usual approach is to estimate emissions from input parameters, such as fuel flow rates, combustion conditions, and equipment standards, or to use proxy output parameters

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[56] The term "emissions" in this section is construed broadly enough to include both air and water pollutants and also solid waste discharges.

[57] Differentiating pollution taxes by sources is also of questionable legality in the United States.

which are assumed to be correlated with emissions, such as flue gas opacity (Blackman and Harrington, 2000). Monitoring is conducted intermittently. As would be expected, these second-best monitoring methods are less accurate, to varying degrees, than direct emissions monitoring.

A ''mass balance'' approach is sometimes used to estimate emissions. This method compares the mass of pollutants in input and output streams, and derives emissions as the difference. When feasible, the mass balance approach provides reasonably accurate emissions estimates (Blackman and Harrington, 2000).

If it is possible to monitor or estimate emissions with reasonable accuracy, the best regulatory design is some form of emissions-based regulation, e.g., an emissions standard, tax, or an emissions-based trading program. As is well known, such a policy design gives the polluter the most options for the regulatory adjustment response, creating the maximum flexibility to minimize abatement costs. This cost-minimizing response will also pass on the ''correct'' signal about the cost of output to consumers, allowing for the optimum demand-side response. For example, a tax on sulfur emissions from coal-fired power plants provides plant managers an incentive to compare the costs of the following combination of options: using an FGD scrubber system to remove the emissions *ex post*, purchasing lower sulfur coal, switching to cleaner fuels such as natural gas, improving fuel economy at the plant or operational efficiency otherwise, or lowering plant output. This flexibility allows the plant manager to choose the least-cost response, or combination of responses, to reduce abatement costs and the price impact of the regulation on consumers.[58]

However, it is sometimes too costly to even imperfectly monitor or estimate emissions, such as those arising from some kinds of nonpoint sources (Shortle and Horan, 2001). It may also be administrative costly to devise emissions-based regulatory systems. In this case, the relevant issue is the best emissions proxy for the regulator to target. Alternative proxies can be closer or further from the point that the emissions are produced in the chain from raw materials supply to final consumption. If emissions are produced from a contaminant in an input — for example, the sulfur content of coal — this containment is the closest emissions surrogate. The least proximate surrogates are complements or substitutes for the output in question,

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[58] This example is taken from Blackman and Harrington (2000).

or complements or substitutes for the inputs. Principles for taxing complements or subsidizing substitutes are based on the demand system in which they occur (Eskland and Jimenez, 1992). An example of this strategy would be to subsidize mass transit systems, a substitute for urban driving, as a way to reduce the congestion, air pollution, and accident externalities associated with vehicular travel. The same principles would apply on the input side: instead of monitoring the nonpoint source emissions of toxic pesticides from farms, the use of more benign pesticide substitutes could be subsidized. Of course, a tax on the pesticide itself would be closer to the source of emissions.

Any departure from directly targeting emissions imposes higher abatement costs, by creating market distortions, and eliminating the incentive polluters have to consider one or more pollution control options which might otherwise be cost-effective. The greater the departure the greater the abatement costs. For example, imposing a tax on the sulfur content of feedstock coal would incentivize a comparison of all the alternatives previously described, except one: the cost of the *ex post* emissions control. On the other hand, imposing a tax on electricity would only provide an incentive to lower output and consumption.

While targeting regulatory instruments away from the point of emissions production raises abatement costs, this strategy can lower administrative costs (Vatn, 2001). To reduce administrative costs, it is best to target points with relatively few actors. For example, the costs of designing and administering a broad-based carbon tax on primary energy producers and importers would be lower than an equivalent tax applied on coal users and electricity consumers throughout the economy. In this particular case, the abatement cost and environmental trade-off would be relatively low, given the uniformly-mixed nature of carbon emissions, the stochiometric link between carbon and CO_2 emissions, and the high cost of post combustion removal and sequestration of carbon emissions. In other cases, targeting a point early in the raw materials-to-consumption chain might reduce administrative costs while imposing a relatively high trade-off in abatement and/or environmental costs.

In summary, broadening the analysis to transaction costs during the policy's operational stage introduces many complex policy design trade-offs. It may be necessary to develop policies with differing mixes of administrative costs, monitoring and enforcement costs, abatement costs, and environmental goals to optimize performance in terms of the policy's overall cost and level of pollution control.

6.1.3 Transaction costs of using environmental revenues

It was mentioned earlier that public revenues can be viewed as a common property resource, and that stakeholders have incentives to attempt to influence the disbursement of public monies. This issue is assessed in a large general literature on rent-seeking (Tullock, 1988). In the environmental policy context, the implication is that entitlements structured to raise revenues will ultimately impose additional transaction costs relative to freely-granted entitlements (see Lee, 1985; Migué and Marceau, 1993), if the rules governing the disposition of revenues are not decided at the time of the policy-making. If the rules are decided at the time of the policy-making, the transaction costs associated with the revenue disbursement will be shifted to the policy-establishment period.

A related issue is whether revenues raised through environmental policy-making will be used for productive purposes, such as to cut labor taxes, or to make investments which would pass a benefit-cost test. If governments are inefficient or corrupt, the value of revenue will be dissipated. If the government is above its optimal size, then reducing revenues provides additional gains, rather than costs, and there would be an added dividend to environmental policies which do not raise revenue (Farrow, 1999). This issue was likely to have been salient in the period following the collapse of the former Soviet union, since down-sizing government through privatization was a major policy objective in this transitional period. It is also likely to be relevant in many less-developed countries with large public sectors. Of course, the size of government, and its efficiency, is a policy issue in more developed countries as well.

6.2 Instrument Choice

This section explores the relationship between environmental policy instruments and transaction costs during the policy's operational phase, holding policy design constant. The first topic is whether monitoring or enforcement costs differ systematically between incentive-based instruments and more prescriptive approaches, either reinforcing or countervailing the conventional normative view based on the standard assessment of the instruments' abatement costs. Because neither incentive-based instruments nor prescriptive policy approaches are homogenous as categories, and in particular, vary within categories along dimensions affecting operational transaction costs,

the impact of instrument-specific attributes are also assessed. Next, trading transaction costs associated with emissions trading programs are considered. Finally, the normative implications are assessed for conventional instrument distinctions when instruments are targeted at differing points in the chain from raw materials extraction to final consumption.

6.2.1 Monitoring and enforcement of incentive-based instruments versus prescriptive regulations

6.2.1.1 Monitoring

There are a number of ways in which the monitoring costs of incentive-based instruments could differ from more prescriptive approaches. One possibility is that the abatement cost-minimizing property of incentive-based instruments itself translates into lower monitoring costs. This possibility was explicitly assessed in a theoretical study by Malik (1992). In his analysis, a firm-specific optimal penalty structure is imposed to insure compliance, and firm-specific standards and monitoring probabilities are chosen to minimize monitoring costs. It turns out that the conditions for minimizing monitoring costs are not the same as for minimizing industry abatement costs. Whether incentive-based instruments are more or less costly to monitor in this model depends on the second derivative of the abatement cost functions.[59] The basic result generalizes to a more realistic model lacking optimal penalties and thus featuring partial compliance.

Considering other dimensions for comparative analysis, it is necessary to distinguish pollution taxes and tradable permits from liability rules and deposit refund systems within the class of incentive-based instruments, since the latter do not require routine emissions monitoring. The lack of monitoring obviously reduces costs, and is considered to be one of the attractive properties of deposit refund systems when it is technically feasible to use such systems (as for controlling the disposal of durable products). Of course the administrative costs of sorting and handling returns can be significant (U.S. EPA, 2001).

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[59] The result is attributable to the fact that the first derivative of the cost functions, embodied in a firm-specific optimal penalty condition, $P = MC/\pi$, enters the cost minimization as a constraint. This constraint gets differentiated in the optimization, so the second derivatives of the cost function define the optimal enforcement strategy.

There is some debate in the literature whether monitoring requirements are higher for taxes and tradable permits than for performance standards, when all instruments target emissions. Anderson *et al.* (1977) argue that there is no significant difference between the monitoring requirements for emissions taxes and prescriptive regulations. Hahn and Hester (1989) hold the same view when comparing tradable permit systems and performance standards. Similarly, Ellerman *et al.*, (2003) argue that EPA's Acid Rain Program could have been effectively monitored using a mass-balance approach, the pre-existing method for measuring emissions before SO_2 regulation was converted into a cap-and-trade program. On the other hand, Blackman and Harrington (2000) argue that incentive-based instruments might be more costly to monitor given that the viability of these policy approaches depends on precise measurement. Pollution taxes will not be paid without accurate monitoring, and the value of rights in permit trading markets, like in any market, requires that the rights definition have content (Blackman and Harrington, 2000). Yet in practice, incentive-based instruments, and in particular, tradable permits, have expanded rapidly, and are used or planned in such challenging contexts as controlling nonpoint source water pollution. For example, annual water pollution loadings from farms are computed using a model based on agricultural production parameters for a nutrient trading system in New Zealand (Rutherford and Cox, 2009). Overall, it seems that the differences in emissions monitoring requirements for emissions taxes, tradable permits, and performance standards are likely to be relatively minor.

In addition to emissions monitoring, the holding of pollution permits or credits must be monitored in emissions trading systems, while for pollution taxes, tax payments must be collected and recorded. However, the use of information technology probably reduces the administrative costs associated with these additional requirements.

Within the class of prescriptive standards, technology-based standards might be theoretically easy to monitor, requiring only the verification of equipment installation. However, the operation, maintenance, and performance of the equipment is not likely to be any easier to monitor than emissions. Moreover, the performance of technology-based standards is often assessed through emissions monitoring. In the United States, technology-based standards are often implemented through statutory language that requires an agency to set standards for emissions or effluents based on some version of ''best technology.'' These regulations are *de facto* technology

standards but the performance measure is emissions, e.g., discharge samples in effluents in the monitoring metric under the Clean Water Act. Vehicle emission inspections programs are another instance where emissions monitoring is used to assess the performance of mandated technology.

6.2.1.2 Enforcement

This section considers alternative instruments in terms of their enforcement costs. To make this comparison meaningful, pollution taxes and tradable allowance programs must be considered separately from liability approaches within the class of incentive-based instruments.

It seems likely that enforcement costs for pollution taxes will be lower than for more prescriptive regulations. When pollution taxes are monitored well enough to function as intended, the tax rate gives an important piece of information: the marginal abatement cost of every firm in the industry. This variable is crucial to determining the sanctioning regime for deterring violations. Moreover, with abatement costs of all polluters equated at the margin, only one penalty level is required to do the job. Thus, the administrative costs of devising a sanctioning regime to efficiently deter pollution violations should be significantly lower for emissions taxes than for prescriptive regulations, all else constant.

Of course, the same principle applies for tradable permit programs, provided they are functioning efficiently (Stranlund *et al.*, 2002). On the other hand, high transaction costs which constrain emissions trading will require quite complex penalty differentiation (Chavez and Stranlund, 2004). But to the degree that there is any trading at all in cap-and-trade programs, marginal abatement costs will be closer to convergence than with emissions standards without any trading. Thus, it cannot be more costly to devise efficient sanctioning mechanisms in tradable permit programs than in an analogue policy imposing standards without the option to trade emissions allowances.

Another reason to think that emissions taxes and tradable permits will impose lower enforcement costs is the lower probability of legal challenges to enforcement actions, at least in the context of policy-making in the United States where such litigation is common. Polluters are less likely to challenge enforcement actions if they regard the rules to which they are subject as reasonable, and rules with the flexibility for economic adjustment are likely to be viewed as more reasonable than more prescriptive standards. Also, judges

are less likely to respond to legal challenges against enforcement actions involving the payments for emissions taxes or tradable permits, which are commonly regarded as the cost of doing business, as opposed to compliance violations with administratively or legally imposed prescriptive regulations, the reasonableness or legal basis for which might be questioned (Malik, 1992). Within the class of prescriptive regulations, legal challenges to enforcement actions are likely to be relatively higher for more prescriptive technology-based standards than for emissions standards.

A final advantage for tradable permits in particular is that firms on the supply side of the market are likely to support effective monitoring and enforcement, because higher compliance rates increase permit prices. Hence, the economic interests of firms in the role of net-sellers are congruent with the compliance objectives of regulators (Ellerman, 2006).

In contrast to pollution taxes and tradable permits, legal liability rules are likely to yield relatively high enforcement costs. Pollution taxes and tradable permits are usually enforced administratively, while legal liability involves enforcement through court system. Menell (1991) considers the use of liability rules for environmental protection to be inefficient. Law suits can impose high transaction costs while failing to provide consistent levels of compensation, or effectively deterring risky behavior.

Enforcement costs increase when sanctions are related specifically to the polluter's actions, as for example, when liability-based policy approaches rely on a negligence standard. Since lack of understanding of rules and procedures, equipment malfunctions, and random environmental fluctuations can lead to violations, *ex post* investigations are required to determine the polluter's degree of culpability.

Moreover, environmental liability schemes require the valuation of environmental damage for which compensation is being sought. This process is often challenging and controversial, as illustrated by legal actions associated with the Oil Protection Act and CERCLA.

Liability, particularly extended strict liability such as that used in CERCLA, can also increase the private costs of carrying out business. These indirect transaction costs have been manifested in the increased costs and limited availability of insurance for corporations and municipalities engaged in potentially damaging activities (Menell, 1991). Extended liability increases the uncertainty associated with the cost distribution of environmental remediation, and distorts relationships between firms and increases their transaction costs by making asset heavy ''deep pocket'' firms

reluctant to contract with "shallow pocket" firms (Boyd and Ingberman, 1997). In these situations, firms are not able to simply choose contracts based on price and instead have to seek out additional information in order to mitigate their financial risk.

6.3 *Transaction Costs of Emissions Trading*

Transaction costs are incurred when emission rights are exchanged in trading programs, and this topic has received significant attention in the literature (e.g., Hahn and Hester, 1989; Stavins, 1995; Kerr and Maré, 1998; Solomon, 1999; Gangadharan, 2000; Colby, 2000; Cason and Gangadharan, 2003; Ellerman *et al.*, 2003, Ellerman and Joskow, 2008). As with any other kind of market activity, the transaction costs of exchanging emission rights arise from the need for market information, the search for trading partners, the negotiation between buyers and sellers, and the execution and certification of trades. Small number of participants and low liquidity are associated with high trading transaction costs and low trading volume, while low transaction costs allow firms to capture the maximum benefit from trades, increasing the scope and volume of trading.

  The transaction costs of exchanging emission rights can be differentiated by the type of program. Trading programs based on credits for emissions reductions are associated with relatively high trading transaction costs, compared to the ideal standard of a liquid, high-volume market in which transaction costs compose a relatively minor part of the market price. As noted in Section 3, emissions trading programs established by EPA in the mid 1970s were of this type (see Hahn and Hester, 1989; Tietenberg *et al.*, 1998; Solomon, 1999; Ellerman *et al.*, 2003), as are various programs now in existence to encourage the international exchange of CO_2 emissions rights, including JI and CDM (Michaelowa *et al.*, 2003; Ellis *et al.*, 2004; Michaelowa and Jotzo, 2005; Antinori and Sathaye, 2007). Also falling into the category of a credit reduction programs are policies now being established to help achieve end-use energy efficiency standards. These standards exist or are pending in a number of states in the U.S., and credited end-use energy savings ("white tags") are tradable in some cases (Wade and Buchner, 2008). In the EU, end-use energy conservation is official policy, and credits for end-use energy savings (white certificates) are tradable, with markets starting to develop (Bertoldi and Huld, 2006). Since 2003, New South

Wales and the ACT in Australia have also participated in a regional market for tradable energy efficiency certificates (Crossley, 2008).[60]

Trading transaction costs are relatively high in these types of programs for several reasons. Emission reductions must be certified against an unobservable counterfactual baseline, requiring the pre-approval of trades to try to minimize the grant of credits for emission reductions which would have occurred anyway (Stavins, 2002). Attempting to validate emissions reductions is a challenging task given information asymmetries between regulators and polluters (Benthem and Kerr, 2010).[61] The pre-approval process imposes transaction costs on the certifying body, the buyers and sellers, and the brokers involved in the exchange. In the case of CO_2 emissions, reductions are project-specific and often generated at downstream points, raising transaction costs. The heterogeneous nature of activities which can give rise to the credits increases transaction costs. *Ex post* verification is required to attempt to validate emission reductions. That activity also generates transaction costs.

The costs of trading emissions reduction credits can be reduced by the standardization of baselines and verification protocols, though these procedures do not necessarily assure better outcomes in terms of actual emissions reductions. Such standardization has been incorporated into some of the recent trading programs for energy efficiency certificates, e.g., the Australian program just mentioned. There has been less success in standardizing baselines and verification protocols for the project-based trading mechanisms now being used to exchange CO_2 emission rights. Conceivably, this problem could become relatively less significant over time as the market matures.

The relatively low volume and fragmented nature of trading activity for emissions reduction credits reduces the incentive for financial intermediaries to participate in the market. Lacking favorable access to financial markets increases risks and the cost of risk management strategies (Ellerman *et al.*, 2003). For example, buyers of CO_2 emissions reduction credits under the CDM must self-insure against risk (Antinori and Sathaye, 2007).

These problems are largely avoided in cap-and-trade programs and trading programs based on "rate averaging, like EPA's lead trading program

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[60] The Australian market is the first of this kind to be instituted in the world.
[61] "Additionality" is the term used in this literature for a *bona fide* incremental emissions reductions from a credible counterfactual baseline.

(see Ellerman *et al.*, 2003).[62] Allowances are clearly defined, avoiding the need for costly precertification. Having clearly-defined, homogenous commodities is a precondition for the development of relatively sophisticated market institutions (Ellerman *et al.*, 2003). Of course, developing new market institutions is not a trivial task; the start-up costs can be high (Colby, 2000). Moreover, trading transaction costs might be expected to be relatively high in the initial phase of the market development. However, experience shows that trading programs begin to operate efficiently relatively quickly, within one to three years, and market transaction costs decline to the point that the "law of one price" is rapidly observed (Ellerman *et al.*, 2003; Ellerman and Joskow, 2008). Typically, over-the-counter trading is the first trading platform to emerge, but organized exchanges also develop and begin to be used, with varying frequency.[63] Market transactions are recorded in electronic registries. Financial intermediaries enter the market, allowing for the development of sophisticated risk management strategies (Ellerman and Joskow, 2008). These institutional features lower trading transaction costs and increase the scope and volume of trading.

A noteworthy feature of emissions trading of any type is the possibility of some correlation between source heterogeneity and abatement cost heterogeneity, and between source heterogeneity and trading transaction costs. If so, there is a relationship between trading transaction costs and abatement cost heterogeneity. That association has the potential to attenuate some of the cost savings achievable through emissions trading. This issue is likely to be most significant for trading programs based on emissions reduction credits. For example, the heterogeneity of sources in less-developed countries generating CO_2 emission reductions offers scope for significant abatement cost savings, but this heterogeneity also raises the costs of acquiring market information and searching for trading partners. Different institutional contexts in different countries also complicate the cost of negotiation and bargaining, and the certification and verification of emissions reductions. These high transaction costs have the potential to offset

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[62] Rather than capping lead rights at an absolute level, a relative baseline (units of lead/gallon) was used. This baseline is conceptually the same as the relative baseline used in the Swedish REP program.

[63] Emissions trading in the U.S. is usually conducted over the counter, but around a third of the trades in the EU ETS were conducted through organized exchanges by 2007 (Ellerman and Joskow, 2008).

some of the abatement costs savings achievable in this context (Michaelowa and Jotzo, 2005).

Markets regulated through cap-and-trade and averaging-type programs might also exhibit some correlation among source heterogeneity, abatement costs variation, and trading transaction costs, but since trading transaction costs are relatively low, the degree to which they attenuate cost savings is relatively less significant. Heterogeneous sources have probably raised market transaction costs in the RECLAIM program, for example, but trading volume is still relatively high. Source heterogeneity was also a key factor in the transaction costs associated with EPA's lead trading program. Transaction costs were relatively high for smaller refineries; refineries which were part of small companies; and refineries lacking trading partners in the same company (Kerr and Maré, 1998). Notwithstanding this fact, abatement cost differences in the industry stimulated significant trading volume and cost savings during the program's operational period (Hahn and Hester, 1989).

6.4 Evaluation of Inconsistently-Targeted Instruments

In the presence of transaction costs, it might be necessary to target different policy instruments at different points in the chain from raw materials supply to final consumption. This distinction must be reflected in a comparative analysis of the instruments. To illustrate, consider the hypothetical problem of regulating particulates from an industrial plant which uses coal as a fuel, when it is not feasible to monitor or estimate the emissions. Suppose there are two feasible policy options: to mandate the installation of an ash removal system, or to tax the firm's output. In this case, it is not clear whether the less-precisely targeted "incentive-based" tax instrument is more or less cost-effective than the more precisely targeted technology standard. The imprecise targeting of the tax has the same kind of effect on abatement costs as a technology-mandate: it reduces the number of channels for cost-minimization, and creates market distortions.

In sum, comparisons of instruments should be made cautiously when transaction costs force them to be targeted non-symmetrically. For the usual normative conclusions to hold, instruments must be targeted at the same point in the chain from raw materials supply to final consumption. The point in the production process at which instruments are targeted is thus another policy design detail which must be normalized in a comparative assessment.

7   The Implications of Transaction Costs for Optimizing Pollution Control, and Assessing Net-Benefits

The previous three sections have assessed how the design and choice of policy instruments affect transaction costs incurred in each of the three stages of the policy process. That analysis was conducted holding the level of pollution control approximately constant, as well as the exogenous (or partially exogenous) factors potentially affecting transaction costs which were discussed in Section 2. The focus now shifts to the optimal level of pollution control when the level of pollution itself becomes a choice variable. In this discussion, the policy design and instrument choices are held constant, as are the other exogenous (or partially exogenous) factors potentially affecting transaction costs. On these assumptions, the implications of transaction costs for benefit-cost analysis are also considered.

7.1   *Determining the Optimal Level of Pollution Control*

Determining the optimal level of pollution control requires knowledge of marginal damage and cost functions. In a positive transaction cost world and, in particular, under conditions of imperfect information, it is difficult to derive empirical measures of these functions. Hence, as mentioned earlier, one of the immediate consequences of transaction costs is to make the optimization task itself difficult. Positive transaction costs will force imperfect optimization in the level of environmental regulation (relative to the theoretical benchmark in a zero transaction costs world) just as it forces imperfect optimization (again relative to a theoretical benchmark) in the choice of policy designs and instruments.

Taking the theoretical perspective, it would seem intuitive that transaction costs should reduce the optimal level of emissions abatement. This will indeed be our ultimate conclusion. However, some important qualifications should be considered in advance. In particular, the default intuition is based on the implicit assumption that policy-makers do not use transfer payments to reduce the incentive firms have to oppose the policy *ex ante*, or to attempt to avoid compliance *ex post*.

The previous sections showed that policy design can be used to attenuate the link between some categories of transaction costs and the level of pollution control. In Section 4, it was seen that firms will actually gain from more stringent environmental regulation if granted full use entitlement to the

environment. More stringent environmental regulation increases the value of this entitlement, so firms earn more compensation as regulations become more stringent. In theory, this should reduce polluters' political opposition to environmental regulation, thus breaking the link between this category of transaction costs and the level of pollution control.

In addition to incentivizing political acceptance, financial incentives could be used as a stick to encourage regulatory compliance. As discussed in the previous section, the classic theory of enforcement proposed by Becker (1968) uses the penalty structure, rather than costly monitoring, to induce compliance behavior. Since compliance with increasingly stringent regulations could in theory be achieved just by raising the level of penalties, more stringent regulatory objectives could in theory be achieved without increasing *ex post* monitoring costs. This kind of policy design would decouple this important category of transaction costs from the level of regulation.

In reality, practical or political constraints will likely limit the flexibility to use financial incentives. If firms bear a portion of their abatement costs, the incidence of these costs will be increasing with the level of regulation. That relationship suggests that firms will have an economic incentive to increasingly oppose regulations as they become more stringent. And if marginal abatement costs are increasing in the level of abatement, incentives to evade the policy *ex post* will also be increasing in the level of regulation. Regulators must respond to these realities, implying that both private and public sector transaction costs will be linked to the level of regulation.

The impact of transaction costs on the benefit side must also be considered. Section 6 showed that tradeoffs are sometimes necessary in policy design between the desired environmental performance objective, and the transaction costs of achieving it. The consequence is that the environmental benefits per unit of emission reduction are lessened in a positive transaction cost world. Thus, transaction costs can both reduce the benefits and increase the costs of more stringent pollution control, reducing the optimal level of regulation. Since different kinds of policy designs and instruments impose different levels of transaction and abatement costs, the optimal regulatory level will reflect particular policy choices.

7.2   Transaction Costs and Benefit-Cost Analysis

The economic evaluation of environmental policies should incorporate all benefits and costs, including the transaction costs related to the policy's

establishment, implementation, and *ex post* administration, monitoring and enforcement. This evaluation standard is obviously not common in practice.

In considering a benefit-cost analysis which would incorporate transaction costs, it is useful to distinguish between the *ex ante* and *ex post* evaluation contexts. The *ex ante* context is the standard one for regulatory impact assessment, and in fact, the majority of benefit-cost analyses of environmental projects and programs. For this evaluation context, it is important to explicitly address the implications of the endogenous and probabilistic nature of policy-making. The relevant question is whether the *expected value* of the policy's conventionally-measured NPV will cover its projected decision-making, and other, transaction costs. This evaluation stance reflects the reality that the outcome of political decision-making is uncertain, so that a potentially beneficial environmental policy might or might not be approved after a costly decision-making process. If the policy is not approved, the political costs end up as unrecovered social waste. The possibility of this outcome should be reflected in the evaluation. That is, a kind of political "risk assessment" should be conducted to assess whether the expected benefits of considering an environmental policy proposal are likely to outweigh the expected costs.

The *ex post* evaluation context offers an assessment of environmental policy after the political test has been passed. In an expanded transaction cost framework, the relevant question in this case becomes: could the added resource cost of the political activity, and other transaction costs, have counterbalanced, or exceeded, the net value of a project whose conventionally-measured NPV was otherwise positive, so that society was worse off for having gone through the political process of approving an environmental policy assessed in the conventional analysis to be economically efficient? Or adapting the language of the literature on rent-seeking, could the political activity over the policy's decision-making totally dissipate, or more than totally dissipate, the policy's social rent, i.e. the policy's supernormal social return, as measured by its positive NPV?

The two evaluation contexts can be formalized as follows. For the *ex ante* decision context, the relevant decision standard should be:

$$\Pi(B - C) - TC > 0, \tag{3}$$

where Π is the probability that the environmental policy proposal will pass a political contest, B is the conventional measure of the project's environmental benefits, C is the polluter's abatement costs, and TC is the projected

transaction costs incurred at all stages of the policy process. Rearranging (3) implies the following condition:

$$B/C > 1 + (1/\Pi)(\mathrm{TC}/C) = 1 + (1/\Pi)\theta, \quad \text{where } \theta \equiv \mathrm{TC}/C. \qquad (4)$$

For the expected net benefits of an environmental policy proposal to be positive, its benefit/cost ratio must exceed 1 by the amount indicated in the second term on the right-hand of (4). This right-hand term is the ratio of transaction costs to abatement costs ($\mathrm{TC}/C \equiv \theta$) multiplied by the inverse of the probability that the project will pass the political contest, $1/\Pi$. Suppose that $\theta = 0.5$ and $\Pi = 0.5$. Substituting these values into (4) implies that the benefit/cost ratio would have to be greater than or equal to 2 for the expected net social value of the policy to be non-negative.

For the *ex post* evaluation, Π in (4) can be set equal to 1. Again using $\theta = 0.5$ gives a benefit/cost ratio of 1.5. The evaluation standard *ex post* is less stringent than the *ex ante* standard, but still higher than the usual evaluation metric of standard benefit-cost analysis.

Some combination of good empirical work or formal modeling would be needed to provide convincing estimates of Π and θ. But it is instructive to consider some plausible ranges. Consider a Π range from 0.4 to 0.6, and a θ range from 0.2 to 0.6. The best case combination ($\Pi = 0.6$, $\theta = 0.2$) gives the *ex ante* B/C standard of 1.33, and an ex post standard of 1.2, while the worst case combination ($\Pi = 0.4$, $\theta = 0.6$) gives the *ex ante* B/C standard of 2.5 and *ex post* evaluation standard of 1.6. In short, the evaluation standard required for project justification ranges from marginally more stringent to substantially more stringent than the usual project evaluation criterion.

Note that if conditions (3) or (4) are not satisfied, the potential Pareto criterion which is usually used to normatively justify benefit-cost analysis will not hold. Define $\mathrm{TC} \equiv C1 + C2$, where $C1$ is the sum of the transaction costs incurred by the beneficiaries of the project to promote the project and C2 is the sum of the transaction costs of the polluters, and other non-beneficiary stakeholders, incurred to oppose the project. In the *ex ante* evaluation context, if $\Pi(B - C) - C1 - C2 < 0$, then $\Pi B - C1 < \Pi C + C2$. In short, the expected social gain from the policy's winners ($\Pi B - C1$) is not sufficient to cover the expected losses of the policy's losers ($\Pi C + C2$). In the *ex post* evaluation context this condition reduces to: $B - C1 < C + C2$.

The simple thought experiments of this section are not conclusive, but they are suggestive. It seems plausible that environmental policies commonly

judged to be economically justified might be imposing net resource costs if evaluated more completely. This is an area which deserves further research.

8   Empirical Assessment of Transaction Costs

How empirically significant are transaction costs to the normative assessment? Transaction costs are obviously fundamental in shaping the decision-making context, and determining the feasible sets of policy goals, designs, and instruments. Less fundamentally, transaction costs are incurred to establish, implement, and operate environmental policy, as shown in the previous sections. The relevant empirical question is how the cumulative sum of transaction costs over the course of the policy-making compares to the policy's abatement costs.

Empirical assessments of transaction costs in the environmental literature are relatively patchy and incomplete. Part of the reason is that the standard conceptual framework does not reflect all transaction costs, so the focus of empirical studies is relatively selective. Estimates of political establishment costs is perhaps the most notable category of empirical information lacking on account of researcher selection bias. But the diffuse distribution of transaction costs across policy stages and stakeholders, and the incomplete reporting and recording of transaction costs, also makes empirical assessment difficult. The labeling of reporting categories is also likely to be inconsistent, and span a number of different kinds of transaction costs. Aggregating broadly-distributed and inconsistently-reported and -labeled information obviously poses significant empirical challenges.

In the remainder of this section, the empirical assessment of transaction costs is considered in two contexts. First, the guidelines in regulatory impact assessments to record and reflect transaction costs imposed on jurisdictions, and private actors, are described. Second, the academic literature on transaction costs in the environmental area is assessed. Some short concluding remarks are then offered.

8.1   Administrative Costs of Regulation

In both the United States and Europe, policy-makers are concerned about the administrative costs policy actions impose on private actors and jurisdictions. Concerns about administrative efficiency have lead to initiatives to measure and reduce administrative costs and, to some degree,

incorporate administrative costs within the process of regulatory impact assessment (RIA).

In the United States, routine agency administrative costs are infrequently included in RIAs, perhaps because such costs are difficult to allocate to specific regulations. However, under The Unfunded Mandate Reform Act (1995),[64] administrative costs imposed on state and local governments must be recorded in RIAs. The Paperwork Reduction Act of 1995 also requires agencies to measure the paperwork burden imposed by federal rule-making.[65] There does not appear to be a standard cost estimation or measurement protocol for computing administrative costs, with discrepancies among statutes and agency interpretations in the way administrative and related costs are defined and measured (see, for example, U.S. GAO, 2010).

The European Union's ''Better Regulation'' program, an effort to improve the efficiency and transparency of EU policy-making, includes administrative cost reduction as a policy objective. In fact, since 2004, the measurement and reduction of administrative cost burdens has been increasingly emphasized (Radaelli, 2007). In 2007 the European Council endorsed an action program requiring the measurement of administrative costs imposed on private and public stakeholders as a consequence of EU legislation, based on standardized methodology (the ''EU standard cost model''). The goal of this program is to reduce administrative cost burdens associated with regulatory implementation by 25% by the year 2012 (European Commission, 2010).

Although these kinds of policies reflect considerable concern about the administrative and information burdens imposed by regulations, the transaction costs associated with policy-making are not fully measured, or fully reflected in RIAs, in either the United States or the European Union.

8.2 Empirical Assessment of Transaction Costs in the Environmental Policy Literature

The measurement of the size and distribution of transaction costs tends to be quite selective in the empirical environmental policy literature. Typically, a subset of transaction cost categories is measured, and their incidence on particular stakeholders is selectively considered e.g., the costs borne by firms (Antinori and Sathaye, 2007; Jaraite *et al.*, 2010), or by public agencies (McCann and Easter, 1999, 2000). There are a few econometric studies

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[64] 42 USC §4332 (1995).
[65] 44 USC §3501 *et seq* (1995).

in the field which describe the effect of trading transaction costs on market participation rates (Kerr and Maré, 1998; Gangadharan, 2000), or the determinants of transaction costs (Antinori and Sathaye, 2007). However, there is not a robust empirical research agenda comparable to that in the institutional economics field, which assesses the effects of transaction costs on the structure of organizations and institutions.[66] Studies which consider the endogenous determination of transaction costs with other variables, or address self-selection issues — for example, the fact that firms involved in trading CO_2 emissions reductions credits usually self-select into these kinds of programs — are just beginning to appear in the literature (see Millard-Ball and Ortolando, 2010).[67] In short, the empirical assessment of transaction costs in the environmental policy area lacks the kinds of studies common in fields with active empirical research agendas.

Since environmental economists often use the term "transaction costs" to refer to the cost of trading emissions rights (Stavins, 1995), it is not surprising that empirical work has often focused on measuring trading transaction costs (Cason and Gangadharan, 2003; Gangadharan, 2000; Kerr and Maré, 1998). For example, a study of transaction costs in EPA's lead trading program found that transaction costs dissipated 10% and 20% of potential trading surpluses (Kerr and Maré, 1998). A study of the RECLAIM found that the probability that firms would enter the market in 1995, the first year after trading began, would have increased by 32% in the absence of transaction costs. By the following year, the figure had dropped to 12% (Gangadharan, 2000). The rapid decline in transaction costs over these two years was attributed to the development of relationships between buyers and sellers, reducing the costs of finding trading partners.

Assessing the costs of brokerage fees, or equivalently, the spread between supply and demand prices, is another way to gauge trading transaction costs. Using these metrics, the well-known cap-and-trade programs in the U.S. and Europe (the Acid Rain Program, the RECLAIM, NO_x Budget Trading Program, and the EU ETS) can be regarded as efficient (Ellerman *et al.*,

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[66] In this field, the semantic conventions are relatively well established, and the implications of transactions costs for institutional organization and performance have captured the interest of a large group of researchers, spawning a significant empirical agenda (Macher and Richman, 2008).

[67] Millard-Ball and Ortolando (2010) address self-section issues in the context of creating emission reduction credits through the CDM.

2003; Ellerman and Joskow, 2008). Trading transaction costs are nominal in these programs.

The magnitude of different kinds of transaction costs incurred by Irish firms during the first phase of the EU ETS (2005–2007) has recently been assessed (Jaraite *et al.*, 2010). In this study, transaction costs were divided into three categories: the costs of trading; the costs of monitoring and verification; and the costs of the initial program implementation. Consistent with other assessments (e.g., Ellerman and Joskow, 2008), trading transaction costs were found to be insignificant. Monitoring and verification costs varied between about 37,000 Euros for small firms to 198,000 Euros for large firms, which translates into 1.51 and 0.02 respectively per verified emission over the 3-year first phase of the ETS. The costs of implementation reveal the same pattern: relatively low implementation costs in absolute terms for small firms (about 13,000 Euros), and high costs for large firms (about 340,000 Euros), translating into high cost per emission for small firms (0.51 Euros) and low costs for large firms (0.03).[68]

A growing body of research assesses the transaction costs falling on firms who buy CO_2 emissions reduction credits from developing countries, for example, through the CDM (Michaelowa *et al.*, 2003; Ellis *et al.*, 2004; Michaelowa and Jotzo, 2005; Antinori and Sathaye, 2007; Millard-Ball and Ortolando, 2010). The fixed costs associated with acquiring market information and locating credit suppliers tend to be high, but relatively higher for smaller projects than larger projects, translating into declining average fixed costs as the size of the project increases. Variable transaction costs are also found declining with project scale. In a study of 28 projects distributed in Latin America, Asia, and Africa, the elasticity of transaction costs with respect to level of emissions reductions was found to be less than 1 (Antinori and Sathaye, 2007). Considering total transaction costs on a per emission–reduction credit basis, costs ranged between \$0.03 for large projects (greater than 1,477,801 tCO_2) and \$4.05 for small projects (less than 154,014 tCO_2), with a weighted average of \$0.36 per emissions reduction credit.[69] These bounds appear to be consistent with others in this literature, e.g., Michaelowa *et al.*, (2003); Ellis *et al.*, (2004), Michaelowa and Jotzo (2005).

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[68] These averages will be asymptotically declining over time for both small and large firms, since implementation costs are fixed.

[69] Figures in \$2002.

The transaction costs associated with attaining end-use energy efficiency standards have also been assessed in the literature. For example, the transaction costs of energy efficiency projects in the EU have been found to range between 9% and 40% of total investment costs (Mundaca and Neij, 2007). However, the incremental effect that trading makes to the transaction costs associated with achieving end-use energy efficiency standards is likely to be relatively low, so the larger question raised by these numbers is whether end-use energy efficiency itself is the best point to target regulatory intervention. Crossley (2008) points out, for example, that for the energy certificate trading system in New South Wales and Canberra, imposing obligations on generators would have been most efficient. However, that option was not feasible for the regional trading market established, because generators were part of an interconnected national network.

The agricultural economics literature also contains studies of transaction costs, based on one or more of the several components of the broad transaction cost definition used in this article. Transaction costs are usually estimated for a particular policy or program (Carpentier *et al.*, 1998; Fang *et al.*, 2005; Falconer and Whitby, 2000; McCann and Easter, 1999, 2000). Programs to reduce nonpoint water pollution impose transaction cost on public agencies ranging from 38 percent of total costs (McCann and Easter, 2000) to 26 percent (Fang *et al.*, 2005) to between two and five percent (Carpentier *et al.*, 1998).

Few empirical studies include transaction cost estimates at all stages of the policy process. Notably, most empirical studies do not attempt to estimate the costs of initial rights establishment. In some cases (Carpentier *et al.*, 1998), it is pointed out that the inequity of the proposed ''low cost solution'' will likely result in such controversy that policy's feasibility is questionable. However, McCann and Easter (1999) and Thompson (1999) provide measures of rent-seeking costs to influence the allocation of rights during policy enactment. To estimate lobbying costs associated with water pollution control policies in Germany and the United States, Thompson (1999) extrapolates information on the financial stakes and recorded lobbying expenditures over the passage of the North American Free Trade agreement.[70]

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[70] This kind of "cost transfer" is the analogue of a "benefit transfer," and would raise some of the same empirical issues. An alternative would be econometric estimation. See de Figueiredo and Silverman (2006) for an application in the field of education.

McCann and Easter (1999) use an interview based procedure to estimate the transaction costs associated with four different nonpoint effluent control programs and find that, depending on the policy under consideration, rent-seeking costs comprise from under 1 percent to over 22 percent of the total transaction cost estimate.

8.3   Conclusion

In view of the empirical challenge of measuring the transaction costs associated with environmental policy-making, a question is begged: could the existing patchy distribution of empirical information reflect an optimal degree of effort? The answer to that question is no — a selective, and less than optimal, degree of attention to transaction costs is partially responsible for the existing state of information — but posing the question does raise the relevant issue. Given that transaction costs themselves constrain the measurement of transaction costs, how relevant and valuable is transaction cost information for the given decision context? Would transaction cost measures tip the balance for the choice of policy designs or instruments, or for the assessment of a policy's net benefits?

Given the transaction costs of measuring transaction costs, the indiscriminate measurement and recording of previously ignored and unmeasured transaction costs cannot be normatively justified. The value of information must be considered against the cost of its acquisition. The results in Section 7.2 are relevant to this point. Political transaction costs raise the benefit/cost threshold to increase social welfare. But additional research should be able to pin down some reasonable ranges for benefit/cost thresholds necessary for projects to cover transaction costs in different kinds of political contexts, e.g., for more or less controversial environmental rules. Call that threshold X for a particular context. In this particular context, transaction cost measurement would only be necessary in the case of an absolute decision — should the project be pursed or not? — for projects having benefit/cost ratios between 1 and X. Projects with benefit/cost ratios less than 1 should not be considered for any reason, and those with benefit/cost ratios of X or greater should increase social welfare regardless of transaction costs. Of course, there are likely to be many projects with benefit/cost ratios in the 1-to-X range, and for which the information is policy relevant. But there will be some project evaluations for which this information is not germane.

Given the high cost of collecting information about transaction costs, empirically-based modeling strategies might be the best approach for generating transaction cost estimates, when this information is relevant. The empirical estimation of transaction costs is an under-explored area in the field, and deserves more attention.

9   Conclusion

This article has developed a framework for environmental policy evaluation which encompasses all costs at all stages of the policy-making. The approach has emphasized the *ex ante* costs of establishing environmental entitlements, and the *ex post* costs of their administration, monitoring, and enforcement. Since particular policy designs and instruments are chosen to implement and maintain environmental rights, the effect of these choices on transaction costs is an important part of the assessment. Also considered is the way policy alternatives influence trade-offs among different categories of transaction costs, abatement costs, and environmental goals, within and across policy-making stages. The role of transaction costs on the optimal level of environmental policy-making, and the implications for benefit–cost analysis, is also assessed. Finally, the literature on the measurement and estimation of transaction costs is briefly considered.

There are a number of implications which are useful to summarize.

9.1   *Normative Consequence of the Policy Establishment Period*

A key point is the logical inconsistency of ignoring the welfare implications of the political costs of establishing environmental entitlements, and the potential bias to the normative evaluation. The welfare costs of political activity to influence the entitlement structure can be larger than the *ex post* cost differentials conventionally used to justify instrument choices, and larger than the welfare costs of the non-optimal, unregulated state (compared to the theoretical benchmark of a costlessly-achieved regulated alternative) which motivates the policy-making. Ignoring this reality risks biasing the assessment of policy designs and instruments, and the overall assessment of the welfare consequences of environmental policy-making.

Modeling the economic costs associated with the policy's establishment should receive higher priority, in view of their potential normative significance. Since the political costs of establishing policy will endogenously

depend on such parameters as the benefits and costs of the policy, and the policy's entitlement structure, modeling should be tractable, particularly within the context of benefit–cost analysis, which must otherwise provide estimates of benefits and costs. The incremental cost of modeling political costs in this context should be relatively low, conceivably requiring less effort than other parts of the evaluation, e.g., the construction of complex models to estimate environmental effects. Given the potential scale of policy-related transaction costs, these costs would seem as relevant to the normative evaluation as the policy's conventionally measured resource costs.

9.2   Multistage Analysis

This study illustrates the limitation of focusing on outcomes in only one stage of the policy process. There are two fundamental reasons. First, considering all stages in the policy analysis enables a comparison of between-stage tradeoffs of different categories of costs, increasing the scope for optimized policy-making. An exclusive focus on any particular stage reduces insight about the incentives actors face which influence their behavior and impose economic costs across all stages of the policy-making.

Secondly, the conventional focus on efficiency during the policy's operational period implicitly assumes that the normative analysis can be dichotomized. In fact, the political cost of establishing environmental entitlements is another channel through which distributional effects have efficiency consequences. These costs cannot be separated from the overall efficiency evaluation.

9.3   Menu of Policy Options

One cost of the limited focus of the conventional policy evaluation approach is to reduce the menu of policy options. The most notable example is the minimal use in actual policy-making of pollution taxes imposed at high enough levels to affect polluters' behavior. The infrequent use of incentive-based tax instruments imposes an economic cost, given that other incentive-based alternatives are not always feasible, or necessarily normatively preferable (see Nordhaus, 2007 and Metcalf, 2009). The more fundamental evaluation allows for the possibility of structuring entitlements to make incentive-based emission taxes politically possible. As shown in Table 2, the theoretical range of options at policy makers' disposal is large, so the scope for optimized policy-making is greater than often assumed.

9.4   Implications for Instrument Choice

In an expanded analysis which covers transaction costs, instrument choice comparisons must be normalized for relevant structural elements of policy design, so that differences are cleanly attributed to the fundamental properties of the instruments themselves. At the minimum, the entitlement structure of policy alternatives, the scope for implementation flexibility, and the environmental performance measure — emissions, output, etc. — should be held constant. If for practical reasons these factors cannot be held constant, then the sources of the differences among instruments should be clearly attributed to the cause, i.e., to asymmetric policy design. Suppose, for example, it is not practical in a particular context to implement the same entitlement structure with an emissions tax as with a performance standard, so that only an emissions tax of the usual kind, in which no entitlements are granted to the polluter, is feasible. And it is not feasible to implement the performance standard with tradable permits. In making the comparison between these two constrained alternatives, it is reasonable to make the usual distinction about the different effects on aggregate abatement costs. That difference is clearly a property of the instruments. But it is not reasonable to draw the conclusion that emission taxes are less politically acceptable than standards. What is less politically acceptable is a particular entitlement structure.

A general problem with drawing conclusions about the properties of instruments at the theoretical level, even with the recommended normalization, is that interactions among the variables affecting transaction costs may make the levels of the variables significant in actual empirical applications. For example, holding all relevant design attributes constant, the relative implementability of incentive-based instruments versus prescriptive approaches will be affected by the stringency of pollution control and the heterogeneity of firms within the regulated industry. The fact that transaction costs are likely to be endogenously determined within a system also complicates clarity about instrument choice comparisons. This issue has not been addressed significantly in theoretical or empirical research on transaction costs in the environmental policy context.

What, if any, categorical generalizations can be offered about policy instruments in a world of positive transaction costs? First, it should be reiterated that it is precisely because transaction costs exist that incentive-based instruments have their normatively desirable properties.

So as a first approximation, transaction costs provide the justification for incentive-based instruments. In fact, the most exact normalization would standardize the policy content of incentive-based instruments and prescriptive approaches — that is, provide a comparison of prescriptive and incentive-based instruments which both yield a cost-effective distribution of pollution control.[71] In this kind of comparison, it would be a rare prescriptive approach which would impose lower transaction costs and abatement costs than an incentive-based alternative. In short, the standard comparison in the literature between incentive-based instruments and prescriptive regulations is comparing one alternative which "does more" to one which "does less."[72] However, even using the inexact comparison of feasible prescriptive approaches to incentive-based alternatives, and holding other relevant design features constant, incentive-based instruments offer flexibility which should make them more politically feasible and implementable than more prescriptive approaches, as described in Section 5. Differences in monitoring costs between prescriptive and incentive-based approaches are somewhat ambiguous, but not likely to be significantly different, and enforcing emission taxes and tradable permits (as opposed to liability approaches) might impose lower transaction costs, as a default, for the reasons described in Section 6. Differences in administrative costs are difficult to generalize across policy instruments.

Of course, transaction costs can render emissions infeasible as a basis for regulation under any policy approach. Once the second-best world is entered in which some other target than emissions becomes the basis for regulation, the comparison between incentive-based instruments and prescriptive approaches breaks down — unless the point at which regulatory action is targeted is symmetric across instruments. Taxes targeting some input or output related to emissions, rather than emissions specifically, reduces the incentive for polluters to fully utilize channels for reducing emissions, and leads to market distortions — precisely the effect of conventional prescriptive approaches. As such, it is probably best to let go of the semantic labels "incentive-based instruments" and "prescriptive regulations" for uncontrolled, cross-level comparisons in the second-best context where the

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[71] This idea is consistent with the procedure described in Kaplow (1992) for normalizing instrument comparisons between "rules" and "standards."

[72] Within the class of incentive-based instruments, tradable allowance programs with a banking option "do more" than tradable allowance programs without this option, or conventional emissions taxes, enabling cross-period cost-minimization.

regulatory basis is something other than emissions. The relevant issue in this context is what policy, however labeled, imposes the lowest trade-off among transaction costs, abatement costs, and goal attenuation.

9.5   Implications for Instrument Choice Frameworks

Expanded instrument choice frameworks are now appearing with some regularity in the literature (e.g., Fullerton, 2001; Harrington *et al.*, 2004; Goulder and Parry, 2008). These frameworks add a number of choice criteria, such as political feasibility, ease of implementation, monitoring and enforcement requirements, distributional equity, effects on technical change, second best effects (related to tax interactions), impacts on competiveness, and the like. This work adds useful insight about additional attributes of the real world policy context which should be incorporated into the normative evaluation.

However, handling real-world complexity through expanded instrument choice frameworks imposes costs, as well as benefits. First, there does not appear to be a consensus about ways to discuss and consider policy attributes beyond the standard efficiency criterion. Terminology is not consistent across frameworks, nor are criteria necessarily identical. Secondly, as pointed out by Goulder and Parry (2008), the weighting and comparison of multiple choice criteria poses problems. It also seems that these frameworks include more and less fundamental criteria in the same list without discrimination. The normative distinction between distributional equity and efficiency is fundamental, for example, whereas as the distinction between monitoring effectiveness, on the one hand, and implementation feasibility, on the other, reflects a difference between two different components of the efficiency evaluation.

In theory, these issues could be avoided by aggregating and monetizing all components of the efficiency evaluation — transaction costs in Stages 1–3, and abatement costs in Stage 3 — to produce a conventional money metric of the policy's comprehensive costs. In this context, the comparison attributes would be reduced to two: the costs and effectiveness of the policy in question. These attributes could be combined into a cost-effectiveness measure, as is frequent in the health economics field (see Drummond *et al.*, 2005). Compared in this way, for example, one might conclude that a voluntary environmental agreement (VEA) for a particular environmental problem is not as cost-effective as a regulatory alternative, because while the costs of

the VEA are low, the effectiveness is lower yet. Of course, cost-effectiveness ratios have their own kinds of methodology issues. But in general, the conventional approach of rendering apples and oranges comparisons commensurable through monetization should apply to the assessment of a policy's efficiency costs, broadly considered. While this approach may not be fully tractable empirically, given the difficulty of measuring all categories of transaction costs, a step in this direction could be taken with more complete modeling and empirical research.

Overall, the points summarized here reflect a fundamental perspective on policy analysis. Of course, it is a truism that context matters. But fundamentals obviously matter as well, and this point has not been fully exploited in the literature on transaction costs in the environmental policy field. A fundamental perspective would provide the basis for well-motivated empirical research, and more precise policy analysis. That seems like a worthy goal for future theoretical and empirical research.

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