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Analysis

The Political Transaction Costs and Uncertainties of Establishing Environmental Rights

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ABSTRACT

The significance of transaction costs for the analysis of environmental policy is increasingly recognized. This article focuses on one aspect of the topic: the political uncertainty and transaction costs of establishing environmental rights. Our contribution is to model the political process around the rights establishment, and to monetize the associated welfare costs. The model includes both policy-related and political–institutional parameters, including the extent to which environmental rights are shared with polluters; the environmental benefits of the policy; the policy's abatement costs, and the relative political power of polluters and environmentalists. The model is solved to give unique Nash equilibria for the transaction costs of lobbying, and for the probability of the policy's political success. These results are then used to show the degree to which political actions can dissipate the expected economic surplus from environmental policymaking.

1. Introduction

Research on the role of transaction costs in environmental policymaking has increased rapidly over the past decade (Garrick et al., 2013a). This subject was recently reviewed and advanced in a special issue in Ecological Economics.[1] The emerging view extends neoclassical approaches to include insights from behavioral economics and diverse institutional perspectives. Expositions of this broader analysis can be found in McCann (2013), Garrick et al. (2013b), and Marshall (2013).

This article addresses one issue in the large transaction cost literature: the costs and uncertainties associated with establishing the rights to use resources. This is itself a broad topic. Rights are established through regulatory initiatives to improve health, safety, homeland security, and the environment, and also to improve the management of natural resources, such as water (see Crase et al., 2005, 2013; Garrick et al., 2013b; Grafton et al., 2011; Pease, 2012; Shortle and Horan, 2008). International agreements are also required to define rights over resources that span national boundaries, or which are located in regions outside jurisdictional limits (Libecap, 2014). Within this broad scope, this article focuses specifically on the assignment of environmental rights through domestic policymaking.

The political actions required to define domestic environmental rights impose significant economic costs, and create uncertainty about the policymaking outcome (Brewer and Libecap, 2009; Buchanan and Vanberg, 1988; Jung et al., 1995; Zetland, 2009, 2011). Yet, traditional economic evaluations — theoretical or applied — do not monetize the welfare costs of establishing environmental rights. This conventional approach implies the logically inconsistent notion that agents are rational before and after the environmental policymaking, while abandoning self interest in the intervening period when the rights are assigned, or that political competition over the rights assignment is expressed only through transfer payments, such as bribes, that have no economic consequence (see Krutilla and Krause, 2011). The latter view is not the standard one in the large public choice literature that studies political behavior and rent-seeking (see Hillman, 2013), although to our knowledge, the public choice literature does not explicitly monetize the welfare costs of assigning environmental rights. Reflecting on the state of the research in water policy and management — a topic that encompasses both environmental and resource policy issues — Garrick et al. (2013b, pp 196) state: "A full treatment of the political economy of transaction costs in water reform is an important future research opportunity."

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[1] Transaction Costs and Environmental Policy. Ecological Economics 88, 1–262, 2013.

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In this article, we take up the study of the welfare costs of establishing environmental rights. The distribution of environmental rights is under the control of policymakers, and thus qualifies as a policy parameter. Our goal is to study the way this parameter affects political behavior and welfare costs. Our investigation is in the spirit of a recommendation in McCann (2013) that environmental policy design be considered as an instrument to reduce transaction costs.

To address the study objective, we develop a stylized model of a political contest that draws upon insights from the political economy and rent-seeking literatures, including articles by Becker (1983), Tullock (1980), and Hillman (2013). The model incorporates both policy-related and political–institutional parameters. As noted, the distribution of environmental rights is the principal focus, but the model also includes parameters for the environmental benefits of the policy, the policy's abatement costs, and the relative political power of polluters and environmentalists. Unique Nash equilibria are derived for lobbying costs and for the probability of the policy's political acceptance. Simulations are conducted to show how these variables respond to changes in the parameter values. The solutions are then incorporated into an ex ante normative standard requiring that the expected value of the policymaking be non-negative. This metric monetizes the full welfare costs of the policymaking, including both the political transaction costs of the political contestation, and the economic costs of the associated political uncertainty.

To preview the basic result, political transaction costs can be exceedingly high — as much as ten times higher than the policy's abatement costs for the upper bound parameter configurations considered. For the expected value of environmental policymaking to be non-negative, the required benefit–cost ratio can be remarkably high — greater than 96 for the upper bounds assessed. However, distributing environmental rights to polluters will greatly mitigate these welfare costs. Indeed, distributing all of the rights to polluters will eliminate these welfare costs entirely.

The model that generates these results is structured as a one-shot simultaneous move game over a single policy proposal, abstracting from the possibility of repeated interactions, or political exchanges among stakeholders over a suite of policy reforms. The model also abstracts from some important categories of transaction costs, such as those required for monitoring and enforcement actions, and the transaction costs falling on the public sector. The implications of these and other consequential stylizations will be addressed in the article's concluding section. But it is worth pointing out in advance that our analysis raises questions about the common recommendation in the double-dividend literature to fully auction or tax environmental rights e.g., Goulder et al. (1999). This recommendation follows from modeling in a second-best general equilibrium setting in which environmental policy exacerbates preexisting labor or capital tax distortions. Using environmental rents to finance offsetting tax cuts (while maintaining the size of government) will mitigate these efficiency costs, leading to the recommendation that environmental policy instruments be structured to raise revenue. However, the welfare costs of political behavior are not considered in this analysis. Thus it is possible that the economic cost of political actions over environmental policy alternatives could exceed the efficiency benefits of charging for environmental rights (see Krutilla and Krause, 2011). In fact, this possibility has been shown using a model in which policymaking is assumed to be exogenous, but stakeholders are able to rent seek over the environmental rents that the policy generates (MacKenzie and Ohndorf, 2012). We will return to this issue in our concluding remarks.

In the meantime, the next section reviews some literature on the structure of environmental policy and its effect on political behavior, while the following section describes a simple conceptual framework for environmental rights sharing, and how this parameter will be incorporated into the model. A political economy model is then developed and its solution derived. The solution is used to show the effects of parameter variation on the political feasibility of environmental policy actions, the associated political transaction costs, and the expected value of environmental policymaking. The final section of the article considers methodology issues and offers some recommendations for future research.

2. Background and Literature

Environmental policymaking legally defines environmental use rights for different stakeholders, and reveals value for these rights either exogenously, by imposing an emissions tax, or endogenously by defining the level of pollution control. This process will cause polluters to reduce emissions, incur abatement costs, and reveal inframarginal rents on residual emissions. The degree of resistance by polluters to this new situation will depend on the degree to which their newly-defined environmental rights entitlement differs from the status quo ante. Polluters view policies that distribute environmental rights to the regulatory authority as an expropriation of their historical property rights — notwithstanding the legal ambiguity of the status quo before the policymaking clarifies it (Bovenberg, 1999; Raymond, 2003). The distribution of environmental rights to the regulatory authority will also impose concentrated financial losses on polluters. As a result, polluters generally oppose policies that require them to pay for the rights to use the environment, such as auctioned tradable permits or emissions taxes, and in fact, would rather be compensated for the losses incurred to forgo their prior use of the environment. These same factors influence the preference of natural resource users over the allocation of use rights (Colby, 2000; Grafton et al., 2011; Pease, 2012).

In contrast to polluters, environmentalists have traditionally been more concerned about the level of pollution control, and its associated benefits, than the disposition of environmental rents. And the revenue benefits of taxing environmental rents are often too diffuse to generate a public constituency in favor of pollution taxation. These perceptual asymmetries have traditionally allowed policymakers to strike a de facto bargain with polluters, granting them enough environmental rights to keep most or all of the inframarginal rents, in exchange for pollution reductions. This political economy has favored the use of regulatory standards or emissions trading programs with significant grandfathering of the emissions rights, and also the use of environmental taxes in the role of user chargers (to finance pollution control for example), with the rates set too low to deter polluting behavior (see Harrington et al., 2004).

An important line of research has explored whether environmental taxes can be set at high enough levels to deter polluting behavior while sharing enough of the environmental right with polluters to reduce political resistance (see Farrow, 1995, 1999; Pezzey, 1992, 2003). In the first-best context of this literature, inframarginal emissions can be exempted from taxation, or some environmental revenue rebated back to polluters lump sum (hereafter, a "tax-subsidy scheme") without affecting the marginal incentive effects of the policy instrument. In fact, the efficiency effects of pollution taxes with varying degrees of rights sharing are equivalent to emissions trading approaches with varying degrees of grandfathering. This result effectively extends the invariance property of the Coase theorem to include the distribution of environmental rights using either price or quantity-based policy instruments (when the latter are implemented using tradable permits).

A tax-subsidy scheme in Sweden offers an example of this class of policy designs. It raises taxes on point-source NOx emissions enough to incentivize polluting firms to reduce them, while rebating collected revenues back to polluters in proportion to their energy use. Less pollution intensive firms than the industry average receive a subsidy on net, while the others pay a tax — but one that is less than the standard emissions charge. This policy has significantly reduced NOx pollution in Sweden (see Sterner and Isaksson, 2006). Buybacks of fishing quotas in New Zealand and water rights in Australia exemplify similar compensation schemes in the resource management context (see Colby, 2000; Crase et al., 2013; Garrick et al., 2013b).

Our purpose in this article is to show how the reduced political resistance to these kinds of policy designs translates into welfare cost savings. The next section describes the environmental rights sharing concept in more detail, and how it will be incorporated into the model as a policy design parameter.

3. Modeling Environmental Rights Sharing

A conceptual framework has been developed to represent environmental rights sharing (see Farrow, 1995, 1999; Pezzey, 1992, 2003). We use an emissions tax to demonstrate the concept, but an equivalent tradable permit system could be used instead.[2] This conceptual representation is then connected to the parameterization in our analytical model.

3.1. Conceptual Model of Environmental Rights Sharing

Assume that polluters face an exogenous pollution tax that reduces their emissions from e_o to e, imposing abatement costs of C (represented as a positive number).[3] Define $\eta (0 \leq \eta \leq 1)$ as the share of the polluters' original emissions level (e_o) granted to them as an environmental entitlement. This environmental entitlement corresponds to the emissions that polluters are allowed to produce without penalty after the policymaking. In this context, $R = t(\mathrm{e} - \eta e_o)$ is the revenue that the policy raises, with $R > 0$ whenever $\eta < \frac{e}{e_o}$ and $R < 0$ whenever $\eta > \frac{e}{e_o}$ — the latter corresponding to an emissions reduction subsidy. The $\frac{e}{e_o}$ term gives the emissions share remaining after the policymaking as a fraction of the original baseline, e_o.

The full impact of the policy on polluters (φ) is composed of the sum of the financial effect, R, and abatement costs, C, as follows:

$$\varphi = t(\mathrm{e} - \eta e_o) + C = R + C. \tag{1}$$

This formulation is based on the implicit assumption that polluters face an infinitely elastic demand curve in the product market, so that the burden of emissions taxation falls on them exclusively. Implications are discussed in the Discussion and Conclusion.

Differentiating Eq. (1) gives: $\frac{\partial \varphi}{\partial \eta} = \frac{\partial R}{\partial \eta} = -te_o < 0$. That is, the environmental rights distribution will not affect the level of pollution, e, or the policy's abatement costs, C, so that η can be varied between 0 and 1 under the assumption that e and C are constant (see Farrow, 1995, 1999; Pezzey, 1992, 2003). And since e is not affected by varying η, the benefits of the policy will be constant as well. In short, varying η only has the financial effect of changing the revenue that the policy raises.

Fig. 1 illustrates Eq. (1) for different degrees of rights sharing between polluters and an environmental authority. Emissions are indicated on the horizontal axis running from zero on the left hand side to the firm's pre-regulation level, e_o, on the right hand side. The marginal value of the emissions to the polluters (MB) and the marginal environmental damage cost of the emissions falling on third parties (MC) are indicated on the vertical axis. The marginal benefit (MB) and marginal environmental cost curves, which are parametrically varied as MC1, MC2, and MC3, have the usual interpretations. For convenience, the marginal cost curves are allowed to intersect the MB curve at e, and only the segments on the extensive margin are shown. The three marginal cost curves are associated with the three benefit–cost ratios: (B1 + B2 + B3 + B4) / C; (B2 + B3 + B4) / C; and (B3 + B4) / C.[4] It will be assumed throughout the analysis that C is constant, so that changes in benefit–cost ratios are equivalent to changes in the level of benefits, as shown in Fig. 1.

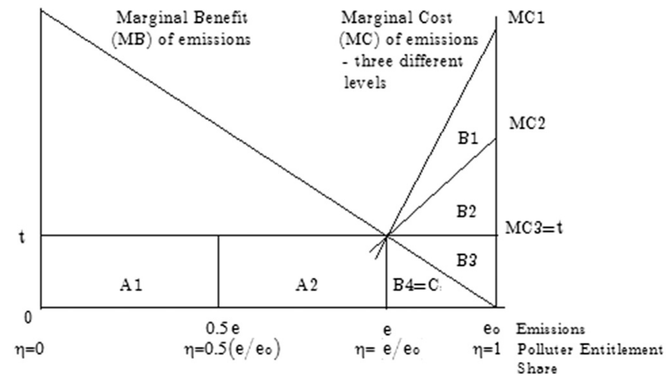

Fig. 1. The structure of environmental policy.

Now consider some cases. As noted above, the environmental authority captures at least some of the environmental rents ($R > 0$) whenever $\eta < \frac{e}{e_o}$. In the limiting case that no environmental rights are assigned to the polluters, $\eta = 0$, Eq. (1) goes to $\psi = te + C$. In this case, the environmental authority captures all of the rents on inframarginal emissions, while the polluters pay te in emissions taxes and incur the abatement costs, C. In Fig. 1, the emissions tax payments are the sum of the areas A1 + A2, and abatement costs are C, so that the polluters' total liability is A1 + A2 + C. This limiting extreme, of course, represents the rights assignment of a conventional emissions tax, and the equivalent Coasean property rights assignment to the environmental authority.

Polluters could be granted a greater-than-zero degree of emissions entitlement on the range $0 < \eta < \frac{e}{e_o}$. Fig. 1 illustrates the case that $\eta = .5\frac{e}{e_o}$, giving polluters an environmental rights entitlement equivalent to $.5e$. This policy could be conceptualized in two ways. First, that $.5e$ inframarginal emissions are exempt from taxation, so that polluters pay only the area A2 in Fig. 1 on the remaining inframarginal emissions above $.5e$. Secondly, that polluters are taxed on all emissions above 0, but the amount A1 is rebated back lump sum — a tax-subsidy scheme.

If polluters are assigned $\eta = \frac{e}{e_o}$ as their rights entitlement, the first term on the right-hand side of Eq. (1) drops out and no revenue is raised, so that the polluters fully capture the inframarginal rents (A1 + A2 in Fig. 1), while incurring the abatement costs, C. This is the rights sharing of the conventional regulatory standard. Finally, to illustrate the case where $\eta > \frac{e}{e_o}$, consider the limiting extreme that the polluters receive their original emissions level as an entitlement, so that $\eta = 1$ — the equivalent of a Coasean property rights assignment to the polluters. In this case, polluters receive an emissions reduction subsidy, $R = t(e - e_o)$ (see Eq. (1)) or, represented as a positive number, the area $B3 + C$ in Fig. 1. The polluters gains on net from this policy by the amount of the area B3 in Fig. 1.

[2] Although emissions taxes and tradable permits are equivalent at the fundamental level considered here, the transaction costs associated with these policy instruments can deviate in practice. New institutional arrangements often are needed for emissions trading systems, requiring additional transaction costs (Colby, 2000; Gomez and Delacamara, 2013; Shortle and Horan, 2008; Zetland, 2009, 2011). Different transaction costs can also be associated with different kinds of trading systems (see Nentjes and Woerdman, 2012). In contrast, pollution taxes can sometimes make use of existing institutional arrangements, e.g., a carbon tax imposed as a markup over an existing sales tax on fuel does not need a new revenue collection system. In this situation, the additional transaction costs associated with the emissions tax may be relatively minor.

[3] To simplify the exposition, polluters are assumed to be homogenous. Implications are discussed in the Discussion and Conclusion.

[4] The letters in the figure denote the areas of the spaces that surround them.

3.2. Representing Environmental Rights Sharing in the Political Economy Model

We now introduce a parameter that will be convenient for representing environmental rights sharing in the analytical model developed in the next section:

$$\gamma \equiv \frac{R}{C} \equiv \frac{t(\mathrm{e}-\eta e_o)}{C}. \tag{2}$$

The γ parameter represents the average revenue the environmental policy raises per unit abatement cost. It might also be described as the policy's "revenue-cost ratio" — the financial analog of the policy's benefit–cost ratio. With *C* assumed to be constant throughout the analysis, increasing γ will correspond to increasing revenue.

From a pure public finance perspective, the higher the value of γ the more efficiently the environmental policy raises revenue, in the sense the lower the economic cost (abatement cost in this context) per unit of revenue raised. However, the γ parameter also indicates the relative financial incidence of the environmental policy on polluters compared to the policy's abatement costs. For example, a value of $\gamma = 2$ indicates both that polluters pay twice as much in emissions taxes as the abatement costs incurred, and also that the environmental authority raises twice as much revenue as the abatement costs. This dual nature of γ reflects the tradeoff between environmental policy designs that raise revenue efficiently in a conventional public finance sense (higher values of γ), and those that will reduce polluters' resistance to the policymaking (lower values of γ).

Eq. (2) shows that γ can take on positive or negative values — the latter again corresponding to an emissions reduction subsidy — with e / e_o the boundary point rights distribution between revenue gains and losses. Because *t*, *C*, *eo* and *e* are constant with respect to rights sharing, differentiating Eq. (2) gives:

$$\frac{\partial\gamma}{\partial\eta} = -\left[\frac{te_o}{C}\right] < 0. \tag{3}$$

The revenue–cost ratio declines as the polluters' environmental rights share increases.

Assume now that the marginal benefits of emissions are linear or can be linearly approximated over the *e*–*eo* range as shown in Fig. 1, so that the abatement costs associated with reducing *eo*–*e* emissions can be approximated as $C = .5\, t(eo - e)$. Making this substitution into Eq. (2) gives:

$$\gamma \equiv \frac{R}{C} = 2\left(\frac{(e/eo)-\eta}{1-(e/eo)}\right). \tag{4}$$

Eq. (4) shows that γ can be expressed as a function of just two parameters, *e* / *eo* and η. The ratio *e* / *eo* on the left-hand side of the bracketed term in the numerator shows the emissions base available for environmental taxation, again expressed as a fraction of the before-policy emissions level, e_o. This term ranges from zero, when the environmental policy reduces emissions to zero ($e = 0$), to 1, when environmental policy has no effect on emissions ($e = e_o$). The η parameter shows how much of this tax base is eroded when environmental rights are distributed to the polluters, with η again measured as a fraction of the original emissions total. The numerator of Eq. (4) reflects the sum of these two parameters, giving what might be labeled as an "entitlement adjusted tax base."

The denominator of (4), $1 - (e / eo)$, shows the degree of the pollution control that the environmental policymaking brings about, with emissions reductions expressed as a fraction of the original emissions level. This emissions range is the base upon which abatement costs are incurred. Eq. (4) is therefore expressing γ as the ratio of an "entitlement adjusted tax base" $(e / eo) - \eta$, to an adjusted abatement cost base, $.5(1 - (e / eo))$ — with the particular adjustment to the latter a result of the linearity assumption.

Table 1
Effects of pollution control and the emissions entitlement on the magnitude of the revenue–cost ratio (γ).

Emissions remaining after pollution control (measured as a fraction of the original emissions level, *e/eo*)	Entitled emissions (η) (measured as a fraction of the original emissions level, *e/eo*)	$R / C = \gamma$
0.91	0.01	20
	0.46	10
	0.91	0
	0.96	−1
0.75	0.00	6
	0.13	5
	0.75	0
	0.88	−1
0.50	0.00	2
	0.25	1
	0.50	0
	0.75	−1
0.00	0.00	0
	0.50	−1

Table 1 indicates γ values that correspond to some selected values for *e* / *eo* and η. The top left-hand cell indicates an environmental policy that incentivizes a relatively modest degree of emissions control, reducing emissions by $1 - (e / eo) = .09$ relative to their original level while leaving the relatively large emissions base, $e / eo = .91$, on which inframarginal rents can be taxed. The first row on the right-hand side shows that $\gamma = 20$ when the environmental authority collects these rents using a conventional emissions tax that distributes virtually no rights to polluters ($\eta = .01$). As more of the rights are distributed to polluters, the value for γ declines. Keeping the level of pollution reduction constant at .09 ($e / eo = .91$), the parameter value $\eta = .46$ corresponds to an emissions tax that collects half of the environmental rents, giving $\gamma = 10$, while the larger entitlement share $\eta = .91$ transfers all inframarginal rents to the polluters, giving $\gamma = 0$. As noted before, this is the rights entitlement of the conventional regulatory standard. When a still greater share of the rights is distributed to the polluters, $\eta = .96$, the regulator is paying an emissions reduction subsidy that exactly compensates the polluters for abatement costs incurred, giving $\gamma = -1$.

Moving down the rows in Table 1, it can be seen in the rightmost column that the maximum γ values are decreasing, corresponding to a decline in the base for environmental rent collection shown in the leftmost column, with *e* / *eo* assuming the successively smaller values of .75, .5, and 0. The level of environmental policymaking places a cap on maximum γ values by determining both the level of taxable inframarginal rents and the magnitude of abatement costs. Of course, the regulatory authority has to capture the rents generated ($\eta = 0$) for the maximum γ value at each abatement level to be achieved. On the other hand, it is possible to achieve the lower γ values in the 0 to −1 range at any abatement level by allowing polluters to capture whatever inframarginal rents exist, or going further and providing compensation to offset abatement costs. This is the general picture from Table 1.

In the simulations in the next section, γ values are allowed to vary from −1 to 20. This range should be thought of as encompassing the kinds of parameter combinations indicated in Table 1. The lower bound is restricted to −1 because complete compensation is sufficient to reduce polluters' incentives for political contestation. The upper bound seems adequate to convey a reasonable minimum for the level of pollution control. The upper bound would be higher for less stringent pollution control (e.g., .05 instead of .09) or for marginal abatement cost curves strictly convex (rather than linear) in emissions control. On the other hand, the maximum γ values would be lower than 20 even for a .09 level of pollution control if marginal abatement cost curves were strictly concave in emissions reduction. Overall, the range −1 to 20 seems a reasonable one to consider in the analysis.

4. The Political Contest Model

In this section, we develop a model of a political contest to represent environmental policymaking. Our conceptual framework is influenced by Becker's political pressure model (Becker, 1983), and is implemented using a one-stage, simultaneous move game similar to those commonly used in the rent-seeking literature (see Hillman, 2013; Tullock, 1980). The model follows the general approach described in Krutilla and Alexeev (2012), but is structured to represent an environmental policymaking process that distributes environmental rights between polluters and an environmental authority, as described in the previous section.

In the model, the environmental authority proposes an environmental policy that polluters oppose and environmentalists support. Both polluters and environmentalists are assumed to be homogenous within their own group.[5] The transaction costs of direct negotiation between the polluters and environmentalists are assumed to be prohibitively high, leaving group lobbying as the only channel to influence the policymaking. In short, the stylization is that of a standard political process. As noted, it is assumed that the polluters and environmentalists act simultaneously, and that this interaction is not repeated in subsequent periods.[6] The goal of environmentalists is to maximize the expected gains from lobbying, while the objective of polluters is to minimize their expected losses. The payoff functions are:

$$\max_{C_1} \pi 1 = B\rho(C_1, C_2) - C_1, \tag{5}$$

$$\min_{C_2} \pi_2 = C(1+\gamma)\rho(C_1, C_2) + C_2. \tag{6}$$

The variable C_1 is the environmentalists' lobbying costs and C_2 is the polluters' lobbying costs. As in the rent-seeking literature, these variables are taken to represent opportunity costs in the conventional sense: the forgone economic value of time and other resources that environmentalists and polluters devote to lobbying. The variables π_1 and π_2 respectively show the environmentalists' and polluters' expected net-pay offs from devoting resources to lobbying. The term $\rho(C_1, C_2)$ denotes a political influence function that gives the probability of the policy's passage as a function of lobbying effort (discussed further below). The exogenous parameters are B, the value of the environmental benefits; C the costs of abatement; and γ, the revenue raised per unit of abatement cost, as described in the previous section. It is assumed that $B > 0$, $C > 0$, $\gamma \geq -1$, and $B > C$.[7] The last assumption implies that the proposed environmental policy is economically efficient in the conventional sense, i.e., that the benefits of the environmental policy are larger than the abatement costs.

The model formulation in Eqs. (5) and (6) implicitly abstracts from the possibility of free riding or other constraints on political action. A qualification is offered in the Discussion and Conclusion. The model is also based on the assumption mentioned in Backgound and Literature that polluters are well-informed about environmental rents, and care about their distribution, whereas environmentalists and the general public do not care about the way environmental rents are distributed. A qualification is offered in the Discussion and Conclusion for the particular case of carbon emissions control.

We use a modified Tullock contest success function to represent $\rho(C_1, C_2)$ as follows:

$$\rho(C_1, C_2) = \frac{1}{1 + \alpha\left(\frac{C_2}{C_1}\right)}. \tag{7}$$

This functional form, and its variants, are widely used in the rent-seeking literature (see Baye et al., 1994; Hillman, 2013; Perez-Castrillo and Verdier, 1992; Van Long, 2013). A variant is also used by Glachant (2005) to study the formation of voluntary environmental agreements. Eq. (7) shows that the probability of the environmental policy's passage, ρ, is inversely related to the lobbying effort of polluters, C_2, and positively related to the lobbying effort of environmentalists, C_1. The α parameter represents the political power of polluters relative to environmentalists, with $\alpha > 1$ ($\alpha < 1$) implying relatively greater (lesser) political power for polluters compared to environmentalists. For example, $\alpha = 2$ implies that 1 unit of lobbying effort by polluters has the same countervailing effect on the probability of the policy's passage as 2 units of lobbying by environmentalists, while $\alpha = .5$ means that 1 unit of lobbying effort by polluters has the same countervailing effect as .5 units of lobbing by environmentalists. Differences in relative political power could arise from political bias associated with "agency capture," or from the weight of legal opinion about the consistency of the proposed environmental policy action with its enabling authority. It is assumed that $\alpha \in (0, \infty)$.

Note that when $\alpha = 1$ and $C_1 = C_2$, Eq. (7) shows that $\rho = .5$. Holding constant $\alpha = 1$, ρ will increase (decrease) from .5 as C_1 is greater than (less than) C_2. Now holding constant $C_1 = C_2$, ρ will increase (decrease) from .5 as α decreases (increases) from 1. In short, the functional form in Eq. (7) captures the effects of lobbying on the outcome of the environmental policymaking in an intuitive way.[8]

Now substituting Eqs. (7) into (5) and (6) and solving for (C_1^*, C_2^*) gives candidates for Nash equilibria. The solutions turn out to be:

$$c_1^* = \frac{\alpha(1+\gamma)\beta^2}{(\beta + \alpha(1+\gamma))^2} \tag{8}$$

$$c_2^* = \frac{\alpha(1+\gamma)^2\beta}{(\beta + \alpha(1+\gamma))^2}. \tag{9}$$

The new variables are defined as $c_1^* \equiv C_1^* / C$; $c_2^* \equiv C_2^* / C$, $\beta \equiv B / C$. The left-hand side of Eqs. (8) and (9) give the ratio of each groups' lobbying costs to pollution abatement costs; the right-hand side includes the exogenous parameters in the model, with benefits and costs combined into the ratio β. It turns out that the solutions in Eqs. (8) and (9) qualify as unique Nash equilibria, since the second order conditions and positive profit conditions hold (see Appendix A).

Note that if Eq. (9) is divided by Eq. (8), the following simple condition results:

$$\frac{c_2^*}{c_1^*} = \frac{C_2^*}{C_1^*} = \frac{1+\gamma}{\beta}. \tag{10}$$

Eq. (10) shows that the ratio of the polluters' to the environmentalists' lobbying costs will be directly proportional to the revenue–cost ratio, γ, and inversely related to the benefit–cost ratio, β. Increasing the revenue raised requires distributing more of the environmental

[5] The implications of this assumption are discussed in the Discussion and Conclusion.

[6] Alternative modeling approaches are discussed in the Discussion and Conclusion.

[7] Whether B and C are thought of as present values, or period values, is not important to the interpretation.

[8] Another variant of the contest success function commonly used in the rent-seeking literature is: $\rho(C_1, C_2) = \frac{1}{1+\left(\frac{C_2}{C_1}\right)^r}$ (see Baye et al., 1994; Hillman, 2013; Perez-Castrillo and Verdier, 1992; Van Long, 2013). In this formulation, the r parameter represents the "returns to lobbying", also described as the "technology of rent-seeking." Increasing its value gives relatively more weight in the decision-making to whichever party is lobbying more. For example, when $r = 0$, $\rho = .5$, whatever the relative lobbying effort. As r goes to infinity, ρ goes to 0 if $C_2 > C_1$, and to 1 if $C_1 > C_2$.

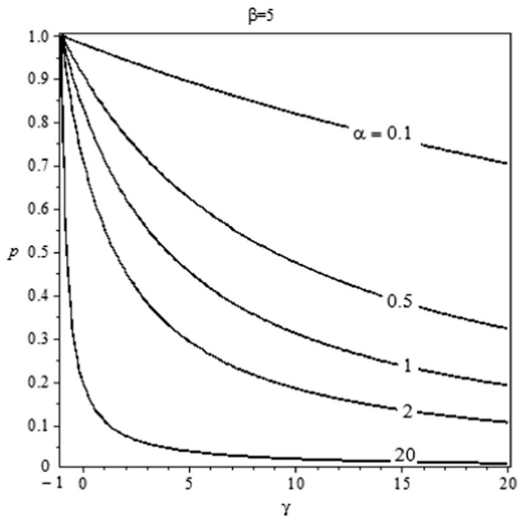

Fig. 2. Probability of environmental policy's political acceptance (ρ) as a function of the policy's revenue–cost ratio (γ).

rights to the regulatory authority, as discussed in the previous section, incentivizing relatively more lobbying from polluters. On the other hand, higher environmental benefits incentivize relatively more lobbying from environmentalists.

Substituting Eq. (10) into Eq. (7) gives the reduced-form probability of the policy's political acceptance:

$$\rho^*(C_1, C_2) = \frac{1}{1+\alpha\left(\dfrac{1+\gamma}{\beta}\right)}. \tag{11}$$

The political feasibility of the policy, ρ^*, is inversely related to the γ and α parameters — the latter increasing in the relative political power of the polluters — and increasing in β.

To give a sense of the magnitudes involved, Fig. 2 shows the political feasibility of the environmental policy as a function of some different parameter values. In the left-hand panel, α is parametrically varied with β fixed at 5. Considering first the middle line at $\alpha = 1$ and at $\gamma = 1$, the probability of the policy's passage is about .71. However, as γ increases to 5, ρ^* declines to .45. Further increasing γ to 10 and 20 reduces ρ^* to .31 and .19 respectively. In short, increasing the revenue that the environmental policy raises significantly lowers the political acceptability of the policy.

Still looking at the left hand panel and taking $\gamma = 5$, it can be seen that $\rho^* = .89$ when $\alpha = .1$, and declines to about .63 when $\alpha = .5$. When α increases further to 2, $\rho^* = .29$. At $\alpha = 20$, $\rho^* = .04$. In sum, increasing the relative political power of the polluters significantly decreases the probability of the policy's passage.

On the other hand, increasing β significantly increases the probability of the policy's passage (right-hand side of Fig. 2). On the assumption that $\alpha = 1$ and again using $\gamma = 5$ to illustrate, it can be seen that $\rho^* = .77$ when $\beta = 20$, but drops to $\rho^* = .63$ when $\beta = 10$, to $\rho^* = .25$ when $\beta = 2$, and to $\rho^* = .17$ when $\beta = 1.2$.

In sum, the α, β, and γ parameters significantly affect the political acceptability of the environmental policy. This is one component in determining the expected net benefits of the policymaking. The other is the political transaction costs that the policymaking elicits. We turn to that topic in the following section.

5. Analyzing the Ratio of Political Transaction Costs to Abatement Costs

The sum of the lobbying cost ratios, $c_1^* + c_2^* \equiv \theta$, gives the ratio of all political transaction costs to the policy's abatement costs. This is obviously an important metric for judging the significance of political transaction costs. If political transaction costs are only a small fraction of abatement costs, the common practice to ignore them will not cause significant biases. But if transaction costs are comparatively large, they should be included in the analysis.

Adding Eqs. (8) and (9) gives:

$$\theta = \frac{\alpha\beta(1+\gamma)(\beta+1+\gamma)}{(\beta+\alpha(1+\gamma))^2}. \tag{12}$$

The signs of the partial derivatives of θ with respect to all parameters turn out to be ambiguous (see Appendix B). To get some sense of the behavior and magnitude of θ, Fig. 3 plots θ for some parameter combinations.[9] Notice first that whenever the entitlement is structured as an emissions reduction subsidy that fully covers abatement costs ($\gamma = -1$), $\theta = 0$ for any combination of the other parameter values. Polluters have no incentive to oppose the policy when they are fully compensated, and the political contest is avoided.

Turning to policy designs with $\gamma \neq -1$, the left-hand panel of Fig. 3 assumes that $\beta = 5$ and varies α as γ increases from -1 to 20. Viewing the $\alpha = 1$ line first, it can be seen that reducing the polluters' entitlement from $\gamma = -1$ to $\gamma = 0$ — that is, completely eliminating the pollution control subsidy — causes θ to rise from zero to .83. Thus, political transaction costs can be non-trivial even when the regulatory authority grants polluters the environmental rights entitlement of the conventional regulatory standard. As the environmental authority turns to revenue-raising policy designs ($\gamma > 0$), θ rises monotonically. For example, $\theta = 1.43$ at $\gamma = 1$ and rises to 2.73 at $\gamma = 5$. When γ increases further to 10, θ goes to 3.44. For $\gamma = 20$, $\theta = 4.02$. The figure indicates that whenever $\alpha \leq 1$, there is a positive relationship between θ and γ, with significant political transaction costs incurred at higher γ values. For example, at $\alpha \leq 1$ and $\gamma = 20$, the value of θ ranges between 3.73 and 5.31. This pattern is observed because increasing the value of γ incentivizes increased lobbying against the policy from polluters, but with $\alpha \leq 1$, the marginal expected payoff to environmentalists from lobbying to promote the policy is also relatively high. The combination of lobbying efficiency for one group and large stakes for the other incentivizes lobbying activity from both parties. Under these circumstances, θ can be several times higher than 1, as indicated in Fig. 3.

The lower two lines in Fig. 3 show the effects of varying γ when polluters are relatively more politically powerful than environmentalists

[9] The parameter values in Fig. 3 are in fact the same as in Fig. 2, except that $\alpha = .05$ in Fig. 3 (rather than .5 as in Fig. 2) to aid the graphical exposition.

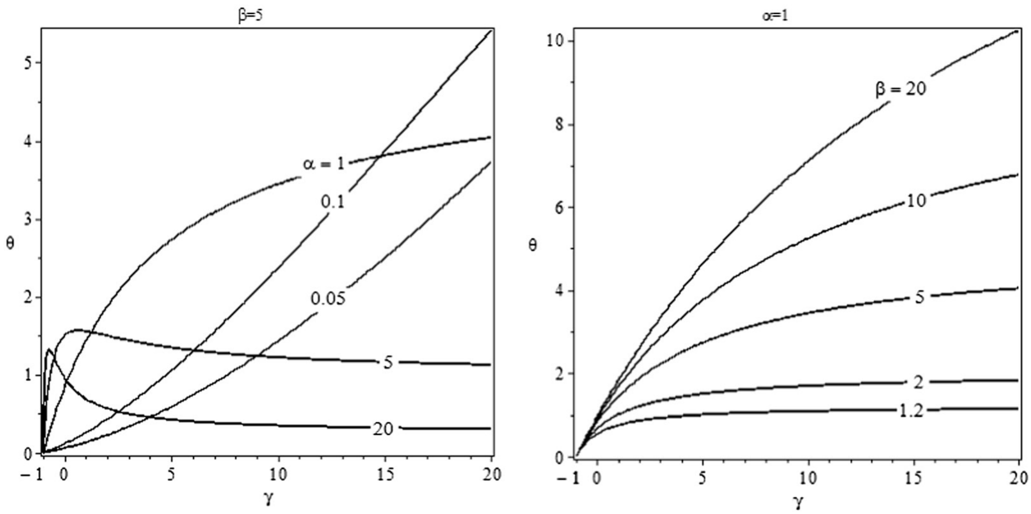

Fig. 3. Ratio of lobbying resource costs to pollution abatement costs (θ) as a function of the policy's revenue–cost ratio (γ).

($\alpha > 1$). Considering first the $\alpha = 5$ line, θ increases from zero to a maximum of 1.56 when γ increases from -1 to 0.67. However, increasing γ further *reduces* political transaction costs. But for any γ value above .67, θ remains greater than 1 (this minimum would change for $\beta \neq 5$). For the bottom line indicating $\alpha = 20$, θ increases from zero to a maximum of 1.32 as γ increases from -1 to $-.72$ Thereafter, θ declines with increasing γ, to a minimum of .3 (at $\alpha = 20$). The bottom two lines in Fig. 3 reflect the fact that a combination of relative lobbying effectiveness for the polluters ($\alpha > 1$) and increasing incentive for the polluters to lobby ($\gamma > 0$) diminishes the incentive for environmentalists to lobby, reducing overall transaction costs.

The right-hand panel of Fig. 3 shows the effects of different values of β on θ for $\alpha = 1$. The value of θ at $\beta = 20$ and $\gamma = 20$ is 10.24. In this situation, polluters and environmentalists have the same degree of political power, and both parties have high stakes in the policy outcome. This combination incentivizes both parties to lobby intensively, giving very high transaction costs. For the much lower benefit–cost ratio of 1.2, the value of θ drops to 1.14. In short, there is an approximately 10-fold difference in θ with a 20-fold in difference in β at $\gamma = 20$.

Variation in β values have less significant effects at lower γ values. At $\gamma = 0$, for example, increasing β from 1.2 to 20 increases θ from .55 to .95. Thus when $\alpha = 1$, distributing the environmental entitlement to the polluters reduces transaction costs significantly over a wide range of possible benefit–cost ratios. Of course, the limiting extreme is granting all entitlements to the polluters, which will reduce transaction costs to zero whatever the size of the benefit–cost ratio.

6. Political Acceptability, Lobbying Costs, and the Overall Efficiency of Environmental Policy

The "political acceptability" of environmental policies is often considered as a qualitative criterion for policy evaluation (see for example, Hahn, 1989; Harrington et al., 2004; Sterner and Isaksson, 2006). In fact, the political acceptability of policies with benefits larger than abatement costs, as is assumed in this article, gives rise to economic value in the sense that the economic surplus that such policies generate has the chance to cover the lobbying transaction costs that the policymaking induces. In contrast, environmental policies that cannot pass a political test will not generate any economic surplus, leaving the political transaction costs over the policymaking as an unrecovered welfare loss.

The reduced-form expressions in Eqs. (11) and (12) can be integrated into a normative metric that enables the welfare effects of political uncertainty and transaction costs to be monetized. The normative standard

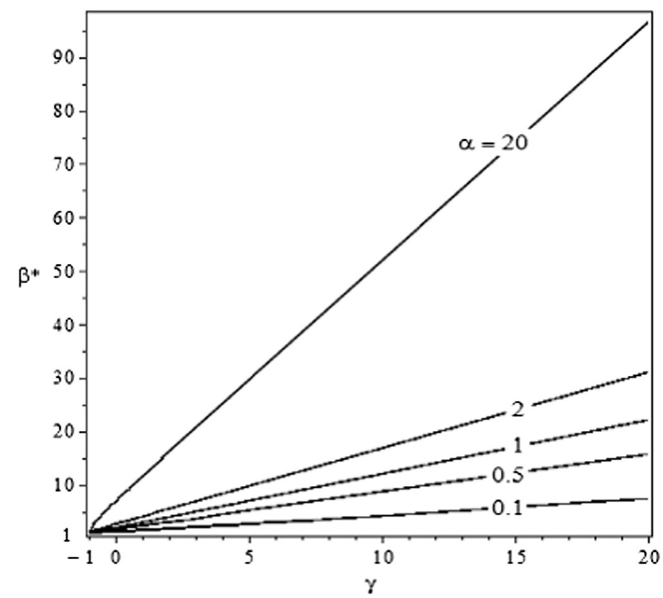

Fig. 4. Environmental benefit–cost ratios required for the expected value of the environmental policy to be non-negative (β^*) as a function of the policy's revenue–cost ratio (γ) and relative political power (α).

requires that the expected net economic value of the environmental policymaking cover political transaction costs, as follows:

$$\rho * (B - C) - (C_1^* + C_2^*) \geq 0. \tag{13}$$

Taking Eq. (13) as an equality, dividing by C, and rearranging gives:

$$\beta = 1 + \frac{\theta(\alpha, \beta, \gamma)}{\rho(\alpha, \beta, \gamma)}. \tag{14}$$

Eq. (14) establishes a threshold ratio for environmental benefits to abatement costs at which the expected net value of the environmental policy just covers its political transaction costs. Solving Eq. (14) for β gives $\beta^*(\alpha, \gamma)$ — the threshold benefit–cost ratio required to meet this criterion.[10]

[10] Pannell et al. (2013) also incorporate measured transaction costs into a benefit–cost standard. The probability of the project's failure is also included in the analysis.

It turns out that the partial derivatives for β^* can be definitively signed, with $\frac{\partial\beta^*}{\partial\gamma}>0$ and $\frac{\partial\beta^*}{\partial\alpha}>0$ (see Appendix C). These signs reflect the negative effect of both γ and α on the probability of the policy's passage (again see Eq. (11)). Although higher values for γ and α do not always increase political transaction costs, as discussed in the previous section, their negative effect on the probability of the policy's passage always dominates in the expected value formulation.

Fig. (4) plots the β^* thresholds against γ for different α values. The relationship between β^* thresholds and γ is linear. The β^* thresholds at $\gamma = 20$ are remarkably high — ranging from $\beta^* = 7.32$ when $\alpha = .1$ to $\beta^* = 96.63$ when $\alpha = 20$. The β^* thresholds are quite high even at lower γ values and more moderate differences in relative political power. For example, at $\gamma = 5$, β^* thresholds range from 5.11 to 9.68 when α changes from .5 to 2. At the level for conventional regulatory standards, $\gamma = 0$, the thresholds range from 1.62 to 2.56 when α changes from .5 to 2. It is only when $\gamma = -1$ that the β^* threshold converges to the conventional benefit–cost standard, $\beta = 1$.

There is a crucial distinction between the α and γ parameters: the α parameter is a characteristic of the political–institutional context over which the environmental decision maker presumably has little, if any, control (at least in the short term), while γ is a policy parameter under the explicit control of the environmental decision maker. The good news is that γ can be used to reduce political uncertainties and transaction costs whatever the characteristics of the political–institutional context. As shown, structuring environmental entitlements to compensate polluters can completely eliminate the welfare costs associated with establishing environmental rights.

7. Discussion and Conclusion

In this research we have modeled the political risks and transaction costs of a political process for distributing environmental rights, and monetized the associated welfare costs. The analysis indicates that the benefit–cost ratios required to justify environmental policy proposals can be far higher than typical — more than 96 for the upper bounds considered — when inframarginal rents are sizeable and emissions taxation is used to fully capture them. However, the distribution of environmental rights significantly affects political behavior and its associated efficiency costs, and this parameter is under the control of environmental policymakers. In fact, the model shows that distributing environmental rights to fully compensate polluters can entirely eliminate the welfare costs of establishing the environmental rights.

There are a number of caveats to offer about the modeling approach that gives these results. For starters, the model does not allow for the kinds of dynamical changes in the institutional environment assessed in McCann (2013), Garrick et al. (2013b), and Marshall (2013). How these factors would affect the direction of the overall conclusions is not clear. Future research would shed light on the welfare effects of institutional evolution in the context of environmental decision-making.

The modeling approach depicts the policymaking process as stakeholders lobbying around a “take it or leave it” policy proposal. This formulation abstracts from the possibility that lobbying and stakeholder negotiation could constructively inform the policymaking, leading to improved policy proposals that are more politically acceptable (see Godwin et al., 2012). As an example, stakeholder involvement has been found to improve investment planning and prioritization to address dry land salinity problems in Australia (Pannell et al., 2013). On the other hand, stakeholder negotiation can also impose transaction costs, and the value of information derived is not always worth these costs (see Crase et al., 2005, 2013). Overall, there are various tradeoffs associated with stakeholder participation in real-world policymaking contexts.

A similar set of issues arises around the possibility that political costs could be reduced through some kind of bargaining process, perhaps by making political compromises over multiple issues, or by negotiating a voluntary environmental agreement. The latter could be formulated as a two-stage game in which the political contest provides the endogenous disagreement point in a second stage that gives the incentives for direct bargaining in the first stage of the game (see Glachant, 2005). To consider this possibility, it is useful to detour briefly into the law and economics literature on civil litigation. A comparison is made there between the costs of stakeholder negotiation in the pretrial stage and the size of the “settlement surplus” — the difference between the most that the defendant would be willing to pay the plaintiff to forgo the legal action and the minimum the plaintiff would accept. If negotiation costs are less than the settlement surplus, there is an incentive to settle out of court. If not, the case goes to trial (Cooter and Rubinfeld, 1989). The direct analog in the environmental policymaking context is whether the negotiating costs of voluntary agreement are higher than the expected surplus derived from avoiding the transaction costs and uncertainties associated with political action. The fact that environmental issues commonly end up as policy conflicts suggests that the transaction costs of direct negotiation are too high for voluntary environmental agreements to be economically feasible in many cases. On the other hand, the existence of voluntary environmental agreements also shows that economic incentives exist to “settle out of the political arena” in other contexts. Exploring how these incentives depend on the parameters in a model such as ours would be a useful area for future research.

The model assumes that polluters cannot pass emissions taxes on to consumers, because the demand for the polluting good is infinitely elastic. If this assumption is relaxed, part of the incidence of emissions taxes would be borne by consumers as a price increase in the product market, reducing the burden of the tax on polluters. However, this price rise would give polluters supernormal returns if they were instead granted the environmental rights (Buchanan and Tullock, 1975). Thus, the impact on the polluters of not being granted the rights will be partly in the form of tax incidence and partly in the form of foregone supernormal returns. Polluters have an obvious incentive to lobby against tax payments, and the rent-seeking literature is premised on the assumption that rational actors will lobby for supernormal returns. Indeed, the excess (short term) profits associated with regulatory standards is one reason put forward in the positive political economy literature to explain why polluters lobby for standards and against emissions taxes (Buchanan and Tullock, 1975). In sum, the compositional balance between tax payments and forgone profits should not affect polluters' lobbying incentives. Thus, the simplifying assumption in our model that polluters' inframarginal losses are exclusively in the form of emissions tax payments should not affect conclusions.

However, the amount of inframarginal rents generated compared to pollution abatement costs, the γ parameter, does matter — as discussed in Analyzing the Ratio of Political Transaction Costs to Abatement Costs — and demand elasticities in the product market will affect the value of γ in ways not represented in our model. For example, the less elastic the market demand for the product, the higher the γ values are likely to be for a given emissions tax, because the price rise in the product market will give smaller output reductions, increasing the ratio of inframarginal rents to abatement costs. In this context, then, relaxing the assumption that the product demand is infinitely elastic might increase political transaction costs and uncertainties. On the other hand, less elastic demands will also shift some of the incidence of abatement costs onto consumers. In contrast to shifting the incidence of rents on the intensive margin, shifting the incidence of abatement costs on the extensive margin will reduce polluters' incentives to lobby. Exploring the implications of these effects in a model that represents market adjustments would be a useful research extension.

A question arises about the consequence of our simplifying assumption that free riding and organizing transaction costs do not reduce political activity, particularly when the membership of lobbying groups is heterogeneous. The transaction costs of organizing political action will obviously reduce lobbying activity to some degree. The question is by how much. Lobbying groups are commonly observed in society, and lobbying actions over environmental policymaking are routine (Harrington et al., 2004). Given this empirical reality, the degree to which free riding and transaction costs reduce lobbying activity is an unanswered question in the political science literature, and an active research area (see Schuler and Rehbein, 1997).

The model shows that the political–institutional context can significantly influence the welfare costs of environmental policymaking, with a wide range of possible outcomes. Research is needed to provide better information on the effects of political–institutional factors on the efficiency costs of establishing environmental rights. Studying other variants of the contest success function would likely provide useful information (see Hillman, 2013; Van Long, 2013). Research to clarify empirical ranges for the α parameter would also provide relevant information, given the sensitivity of the results to its value.

The model excludes certain categories of transaction costs, such as the costs of monitoring and enforcement, and the costs to the public sector of political decision making. Transaction costs falling on the public sector are assumed away in the contest success function, which represents the policymaking process as costlessly responding to lobbying pressure. In actual policymaking, of course, the public sector bears significant transaction costs. For agro-environmental policies, McCann et al. (2005) found that public sector transaction costs were on the order of 30% of the total program costs. For an inefficiently administered public program to reduce dry land salinity in Australia, Pannell et al. (2013) found that public sector transaction costs could exceed production costs by more than two times. Contested regulatory policies also exhibit high public sector transaction costs, because governmental agencies are forced to respond to political actions and litigation from organized special interests (see Harrington et al., 2004).

It was noted earlier that the political economy of controlling carbon emissions is likely to represent a special case. There are several reasons why. First, a program to reduce carbon emissions comprehensively by 10% to 30% will generate high γ values, and the magnitude of the rents will dwarf the size of those from conventional environmental policies. Secondly, the price effects of reducing carbon emissions will be visible to consumers and producers, and distributed along a multistage supply–consumption chain, with primary producers, refiners, distributers, utilities, and consumers among others. This means that there are likely to be more parties competing for the rents than for the rents arising from other kinds of environmental policies, increasing political transaction costs and uncertainties. Third, the scope for policy design to minimize these effects could be relatively constrained, because there are not enough rents to distribute to all potential claimants (CBO, 2003). To advance knowledge of these issues, it would be useful to bring into the analysis the insights and modeling frameworks from the large public choice literature on rent-seeking.

Apart from the special case of carbon emissions, it is not clear whether the sum of the qualifications offered here will yield net positive or negative biases. And the situational complexity of actual empirical contexts will reduce the scope for generalization in any event. But it does seem reasonable to conclude that the welfare costs associated with establishing environmental rights will be normatively significant some of the time, and that these costs could be substantially higher than the policy's abatement costs. Additionally, the welfare costs of environmental policymaking are likely to increase in many contexts when rents are captured from polluters and used for public finance. That suggests the need to consider whether the value of public revenue generated in this way is worth the extra cost. As noted before, this possibility has not been considered in the double-dividend literature, which generally recommends the use of revenue-raising environmental policy instruments. However, it seems plausible that other forms of revenue raising — small increases in a broad-based sales tax for example — might generate less political resistance than transferring environmental rights from polluters. Raising public revenue with non-environmental instruments would offer the benefit of preserving a degree of freedom in the design of environmental policy. That flexibility could be used to reduce political uncertainties and transaction costs arising from policymaking.

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Appendix A. Nash Equilibria for Lobbying Costs

The first order conditions (FOCs) for the beneficiary and the loser, respectively, are:

$$\frac{\partial}{\partial C_1}\left[B\left(1+\alpha\frac{C_2}{C_1}\right)^{-1}-C_1\right]=0 \tag{A.1}$$

$$\frac{\partial}{\partial C_2}\left[(-\gamma C-C)\left(1+\alpha\frac{C_2}{C_1}\right)^{-1}-C_2\right]=0. \tag{A.2}$$

Denoting $\frac{C_1}{C}\equiv c_1$, $\frac{C_2}{C}\equiv c_2$, $\beta\equiv\frac{B}{C}$, the FOCs (A.1) and (A.2) take the form:

$$\frac{\partial\pi_1}{\partial c_1}=\frac{\alpha\beta\frac{c_2}{c_1}}{c_1\left(1+\alpha\frac{c_2}{c_1}\right)^2}-1=0 \tag{A.3}$$

$$\frac{\partial\pi_2}{\partial c_2}=\frac{\alpha(1+\gamma)\frac{c_2}{c_1}}{c_2\left(1+\alpha\frac{c_2}{c_1}\right)^2}-1=0. \tag{A.4}$$

Solving the FOCs (A.3)–(A.4) with respect to (c_1, c_2) gives uniquely defined solutions for the set of parameters $\{\alpha, \beta, \gamma\}$ as follows:

$$c_1^*=\frac{\alpha(1+\gamma)\beta^2}{(\beta+\alpha(1+\gamma))^2} \tag{A.5}$$

$$c_2^*=\frac{\alpha(1+\gamma)^2\beta}{(\beta+\alpha(1+\gamma))^2}. \tag{A.6}$$

Second derivatives of Eqs. (A.3) and (A.4), respectively, are

$$\frac{\partial^2\pi_1}{\partial c_1^2}=\frac{-2\alpha\beta\frac{c_2}{c_1}}{\left(1+\alpha\frac{c_2}{c_1}\right)^3{c_1}^2}<0 \tag{A.7}$$

$$\frac{\partial^2\pi_2}{\partial c_2^2}=\frac{-2\alpha^2(1+\gamma)\left(\frac{c_2}{c_1}\right)^2}{\left(1+\alpha\frac{c_2}{c_1}\right)^3{c_2}^2}<0 \tag{A.8}$$

and since $c_1^*>0$ and $c_2^*>0$ at $\gamma>-1$, the relationships (A.5) and (A.6) define maxima.

Both payoff at (c_1^*, c_2^*) are strictly positive. The payoff $\pi_1(c_1^*, c_2^*)$ is:

$$\pi_1(c_1^*, c_2^*) = \frac{\beta}{\left(1+\alpha\left(\frac{1+\gamma}{\beta}\right)\right)^2} > 0. \tag{A.9}$$

The payoff of the regulatory opponent requires $\pi_2(c_1^*, c_2^*) - (1+\gamma)$, or:

$$\pi_2(c_1^*, c_2^*) = \frac{-(1+\gamma)\left(1+2\alpha\left(\frac{1+\gamma}{\beta}\right)\right)}{\left(1+\alpha\left(\frac{1+\gamma}{\beta}\right)\right)^2} \geq -(1+\gamma).$$

Simplifying yields:

$$\frac{(1+\gamma)\left(\alpha^2\left(\frac{1+\gamma}{\beta}\right)^2\right)}{\left(1+\alpha\left(\frac{1+\gamma}{\beta}\right)\right)^2} > 0. \tag{A.10}$$

In sum, because the second order conditions and positive profit conditions hold, Eqs. (A.5) and (A.6) qualify as unique Nash Equilibria.

Appendix B. Dependence of Political Transaction Cost $\theta = c_1^* + c_2^*$ on (α, β, γ) in the Nash Equilibrium (c_1^*, c_2^*)

For $\theta = c_1^* + c_2^*$ is defined as:

$$\theta = \frac{\alpha\beta(1+\gamma)(1+\gamma+\beta)}{(\beta+\alpha(1+\gamma))^2}. \tag{A.9}$$

The derivative $\frac{\partial\theta}{\partial\gamma}$ takes form:

$$\frac{\partial\theta}{\partial\gamma} = \frac{(\alpha(2-\alpha)(1+\gamma)+\beta)}{\beta\left(1+\alpha\frac{1+\gamma}{\beta}\right)}. \tag{A.10}$$

The sign of $\frac{\partial\theta}{\partial\gamma}$ is ambiguous, and is defined as the following:

$$\frac{\partial\theta}{\partial\gamma} \gtrless 0 \quad \textit{if} \quad \gamma \gtrless \frac{\beta}{\alpha-2} - 1. \tag{A.11}$$

Similarly, the signs for the other parameters are ambiguous:

$$\frac{\partial\theta}{\partial\alpha} = \frac{(\beta+\gamma+1)\left(\frac{1+\gamma}{\beta}\right)\left(1-\alpha\left(\frac{1+\gamma}{\beta}\right)\right)}{\left(1+\alpha\left(\frac{1+\gamma}{\beta}\right)\right)^3} \tag{A.12}$$

$$\frac{\partial\theta}{\partial\alpha} \gtrless 0 \quad \textit{if} \quad \alpha \gtrless \frac{\beta}{1+\gamma} \tag{A.13}$$

$$\frac{\partial\theta}{\partial\beta} = \frac{\alpha\left(\frac{1+\gamma}{\beta}\right)^2(\beta(2\alpha-1)+\alpha(\gamma+1))}{\beta\left(1+\alpha\left(\frac{1+\gamma}{\beta}\right)\right)^3} \tag{A.14}$$

$$\frac{\partial\theta}{\partial\beta} \gtrless 0 \quad \text{if} \quad \beta \gtrless \frac{\alpha(\gamma+1)}{(1-2\alpha)}. \tag{A.15}$$

Appendix C. Dependence of the Threshold Benefit–Cost Ratio β^* on (α, γ) in the Nash Equilibrium (c_1^*, c_2^*)

The threshold benefit–cost ratio β^* is defined as the β value that solves:

$$\beta = 1 + \frac{\theta(c_1^*(\alpha,\beta,\gamma), c_2^*(\alpha,\beta,\gamma))}{\rho(c_1^*(\alpha,\beta,\gamma), c_2^*(\alpha,\beta,\gamma))}. \tag{A.16}$$

The relevant solution of Eq. (A.16) can be written as:

$$\beta^* = \frac{1}{2}\left(1+\sqrt{1+4\alpha(\gamma^2+3\gamma+2)}\right). \tag{A.17}$$

The derivative $\frac{\partial\beta^*}{\partial\gamma}$ can be derived as the following:

$$\frac{\partial\beta^*}{\partial\gamma} = \frac{\alpha(2\gamma+3)}{\sqrt{1+4\alpha(\gamma^2+3\gamma+2)}} > 0. \tag{A.18}$$

Clearly, Eq. (A.13) is always positive in γ.
The derivative $\frac{\partial\beta^*}{\partial\alpha}$ is also positive:

$$\frac{\partial\beta^*}{\partial\alpha} = \frac{\gamma^2+3\gamma+2}{\sqrt{1+4\alpha(\gamma^2+3\gamma+2)}} > 0. \tag{A.19}$$

References

Baye, M., Kovenock, D., de Vries, C., 1994. The solution to the Tullock rent-seeking game when R > 2: mixed-strategy equilibria and mean dissipation rates. Public Choice 81 (3), 363–380.
Becker, G., 1983. A theory of competition among pressure groups for political influence. Q. J. Econ. 98, 371–400.
Bovenberg, A., 1999. Green tax reforms and the double dividend: an updated reader's guide. Int. Tax Public Financ. 6, 421–443.
Brewer, J., Libecap, G., 2009. Property rights and the public trust doctrine in environmental protection and natural resource conservation. Aust. J. Agric. Resour. Econ. 53 (1), 1–17.
Buchanan, J., Tullock, G., 1975. Polluters' profits and political response: direct controls versus taxes. Am. Econ. Rev. 65, 139–147.
Buchanan, J., Vanberg, V., 1988. The politicization of market failure. Public Choice 57, 101–113.
CBO, 2003. Shifting the Cost Burden of a Carbon Cap-and-Trade Program. U.S. Congressional Budget Office, Washington, D.C.
Colby, B., 2000. Cap-and-trade policy challenges: a tale of three markets. Land Econ. 76 (4), 638–658.
Cooter, R., Rubinfeld, D., 1989. Economic analysis of legal disputes and their resolution. J. Econ. Lit. 27, 1067–1097.
Crase, L., Dollery, B., Wallis, J., 2005. Community consultation in public policy: the case of the Murray–Darling Basin of Australia. Aust. J. Polit. Sci. 40 (2), 221–237.
Crase, L., O'Keefe, S., Dollery, B., 2013. Talk is cheap, or is it? The cost of consulting about uncertain reallocation of water in the Murray–Darling Basin, Australia. Ecol. Econ. 88, 206–2013.
Farrow, S., 1995. The dual political economy of taxes and tradable permits. Econ. Lett. 49, 217–220.
Farrow, S., 1999. The duality of taxes and tradable permits: a survey with applications in Central and Eastern Europe. Environ. Dev. Econ. 4, 519–535.
Garrick, D., McCann, L., Pannell, J., 2013a. Transaction costs and environmental policy: taking stock, looking forward. Ecol. Econ. 88, 182–184.
Garrick, D., Whitten, S., Coggan, S., 2013b. Understanding the evolution and performance of water markets and allocation policy: a transaction costs analysis framework. Ecol. Econ. 88, 195–205.
Glachant, M., 2005. Voluntary agreements in a rent-seeking environment. The Handbook of Environmental Voluntary Agreements. 43(2), pp. 49–63.
Godwin, K., Ainsworth, S., Godwin, E., 2012. Lobbying and Policymaking: The Public Pursuit of Private Interests. Cq Press.
Gomez, C., Delacamara, G., 2013. Evaluating economic policy instrument for sustainable water management in Europe (EPI); WP4 ex ante case studies. Final Report.
Goulder, L.H., Parry, I., Williams, R., Burtraw, D., 1999. The cost-effectiveness of alternative instruments for environmental protection in a second best setting. J. Public Econ. 72, 329–360.
Grafton, R., Libecap, G., McGlennon, S., Landry, C., O'Brien, B., 2011. An integrated assessment of water markets: a cross-country comparison. Rev. Environ. Econ. Policy 5 (2), 219–239.
Hahn, R., 1989. Economic prescriptions for environmental problems: how the patient followed the doctor's orders. J. Econ. Perspect. 3, 95–114.

Harrington, W., Morgenstern, R., Sterner, T. (Eds.), 2004. Choosing Environmental Policy: Comparing Instruments and Outcomes in the United States and Europe. Resources for the Future Press, Washington, D.C.
Hillman, A., 2013. Rent seeking. Chapter 19 in The Elgar Companion to Public Choice, 2nd edition. Edward Elgar Publishing, Inc., Northhampton, Massachusetts.
Jung, C., Krutilla, K., Kip Viscusi, W., Boyd, R., 1995. The Coase theorem in rent-seeking society. Int. Rev. Law Econ. 15 (3), 259–268.
Krutilla, K., Alexeev, A., 2012. Normative implications of political decision-making for benefit–cost analysis. J. Benefit Cost Anal. (3: 2) (Article 2).
Krutilla, K., Krause, R., 2011. Transaction costs and environmental policy: an assessment framework and literature review. Int. Rev. Environ. Resour. Econ. 4, 261–354.
Libecap, G., 2014. Addressing global environmental externalities: transaction costs considerations. J. Econ. Lit. 52 (2), 424–479.
MacKenzie, I.A., Ohndorf, M., 2012. Cap-and-trade, taxes, and distributional conflict. J. Environ. Econ. Manag. 63 (1), 51–65.
Marshall, G., 2013. Transaction costs, collective action and adaptation in managing complex social–ecological systems. Ecol. Econ. 88, 185–194.
McCann, L., 2013. Transaction costs and environmental policy design. Ecol. Econ. 88, 253–262.
McCann, L., Colby, K., Easter, W., Kasterine, A., Kuperan, K.V., 2005. Transaction cost measurement for evaluating environmental policies. Ecol. Econ. 52, 527–542.
Nentjes, A., Woerdman, E., 2012. Tradable permits versus tradable credits: a survey and analysis. Int. Rev. Environ. Resour. Econ. 6 (1), 1–78.
Pannell, D.J., Roberts, A., Park, G., Alexander, J., 2013. Improving environmental decisions: a transaction-costs story. Ecol. Econ. 88, 244–252.
Pease, M., 2012. Water transfer laws and policies: tough questions and institutional reform for the western United States. J. Nat. Resour. Policy Res. 4 (2), 103–119.
Perez-Castrillo, D., Verdier, T., 1992. A general analysis of rent-seeking games. Public Choice 73, 335–350.
Pezzey, J., 1992. The symmetry between controlling pollution by price and controlling it by quantity. Can. J. Econ. 25, 983–991.
Pezzey, J., 2003. Emission taxes and tradable permits: a comparison of views on long-run efficiency. Environ. Resour. Econ. 26, 329–342.
Raymond, L., 2003. Private Rights in Public Resources: Equity and Property Allocation in Market-based Environmental Policy. RFF Press, Washington, DC.
Schuler, D.A., Rehbein, K., 1997. The filtering role of the firm in corporate political involvement. Bus. Soc. 36 (2), 116–139.
Shortle, J.S., Horan, R.D., 2008. The economics of water quality trading. Int. Rev. Environ. Resour. Econ. 2 (2), 101–133.
Sterner, T., Isaksson, L.H., 2006. Refunded emission payments theory: distribution of costs, and Swedish experience of NOx abatement. Ecol. Econ. 57, 93–106.
Tullock, G., 1980. Efficient rent seeking. In: Buchanan, J., Tollison, R.D., Tullock, G. (Eds.), Toward A Theory of the Rent-seeking Society. Texas A&M University Press, College Station, Texas.
Van Long, N., 2013. The theory of contests: a unified model and review of the literature. Eur. J. Polit. Econ. 32, 161–181.
Zetland, D.A., 2009. Water reallocation in California: a broken hub will not wheel. J. Contemp. Water Res. Educ. 144, 18–28.
Zetland, D., 2011. The End of Abundance: Economic Solutions to Water Scarcity. Aguanomics Press.