Environmental Rights Conflicts and Institutional Choice: An Economic Evaluation of Environmental Policymaking

Kerry Krutilla and Alexander Alexeev
School of Public and Environmental Affairs
Indiana University Bloomington

Presented at the Ostrom Symposium on Natural Resource Governance
Organizers: Dean Lueck, Indiana University
October 7-8, 2016
Ostrom Workshop
Indiana University Bloomington
September 2016

Draft: Not for citation or quotation

Acknowledgements

We appreciate comments and suggested references from Michael Baye, Daniel Cole, and Arye Hillman. Thanks also to Dean Lueck for inviting the paper and organizing the symposium; to Jim Walker, the paper's discussant; to the other symposium participants; and to Lee Alston, the Director of the Ostrom Workshop.

ABSTRACT

This article formulates a stylized model to measure the welfare costs of political conflict over environmental rights disputes. The parameters in the model represent the economic value of policy proposals, and structural characteristics of the political decision-making process, such as its noisiness and responsiveness to lobbying actions. Also included is a parameter for the degree to which environmental rents are visible to non-polluting stakeholders, and the main policy parameter – the share of environmental rents captured by the government sector. The welfare cost measures are used to demarcate a boundary for the economically efficient public governance of environmental resources.

I. Introduction

In an idealized classical view, private markets are expected to emerge when the net benefits of establishing property rights are positive, and the costs of market transactions are low enough to enable trading. When externalities impede this development, a superordinate governmental authority can step in to clarify the rights situation, facilitating market transactions (Coase, 1960).

Bargaining and other transactions costs are frequently cited as constraints on the internalization of externalities through market institutions. In this case, management by the government sector is commonly stated as the most feasible institutional arrangement (Hanley et al., 1997). The research pioneered by Elinor Ostrom has documented the emergence of other kinds of institutions; in particular, for the management of common-pool resources such as water, land, and forests (see Hayes and Ostrom, 2005; Ostrom and Nagendra, 2006).

However, political conflict over property rights sometimes hinders the emergence of governance institutions, or reduces the efficiency of the institutions which do evolve relative to those which do not (Libecap, 1989). This explanation underlies the world-wide dominance of regulatory institutions for the implementation of environmental, health, and safety policies – a less economically-efficient method to reduce externalities than the use of corrective taxes. Of course, transactions costs or other barriers sometimes constrain the use of corrective taxes. But this fact alone cannot explain the widespread dominance of regulatory institutions.[1] As an example, user charges on pollutant emissions are frequently observed but crucially, *at levels too low to affect polluting behavior.* Examples include charges on water dischargers in France to finance sewage treatment, and a several-cent per barrel tax on petroleum in the United States which generates earmarked revenue to finance oil spill cleanup (Barthold, 1994; Harrington et al., 2004). In these situations, transaction costs are not constraining *ipso facto*, begging the question: why are the rates of existing taxes on externalities not raised to levels which would disincentivize the externality-generating behavior? The answer is that low-level charges allow the stakeholders who generate the externalities to capture most of the rents, while raising the tax levels would transfer the rents to other parties. A variant of this political economy is seen in the widespread sectoral exemptions or concessional tax rates granted to CO_2 emitters under the various ecological tax reforms in Europe (see Beuermann et al., 2006; Dresner et al., 2006; Pearce, 2006).

The political economy literature suggests several reasons why regulatory restrictions enabling private rent capture dominant the political landscape. Again using pollution control as an example, stakeholders who generate pollution experience concentrated losses when taxes or auctioned permits transfer rents to environmental agencies (Buchanan and Tullock, 1975). Equity norms reflecting the view that rent transfers constitute an inequitable property rights taking buttress polluters' rational

[1] In practice, whether or not the transaction costs of corrective taxes are lower or higher than those for conventional regulatory instruments is an empirical question (See Cole and Grossman, 2002a; Krutilla and Krause, 2011). These transaction costs arise from the administration, monitoring, and enforcement actions required to implement any kind of policy.

aversion against financial losses (Bovenberg,1999; Raymond, 2003).[2] Supernormal returns can also be earned when regulation restricts entry (Buchanan and Tullock, 1975; Maloney and McCormick, 1982). Entry barriers are often legislated in the form of "new source" standards which are more restrictive than those that apply to existing sources.[3]

Regulators themselves have traditionally been more concerned with the level of regulation than the allocation of rents. Administrative rulemaking generally responds to statutory obligations targeting the level of pollution control; hence, pollution control is regulators' main concern. Political pressure from environmental group has traditionally reinforced this perspective. Environmental improvement is seen as the primary goal, and environmental rents are not generally very visible to non-polluters owing to the difficulty of discerning regulatory price effects amid other price changes in the economy, or observing the effects of environmental-rent sourced budgetary expenditures allocated over large populations. This perceptual asymmetry has allowed bargains to be struck among polluters, regulators, and environmental groups in which environmentalists willingly trade off rent capture in exchange for polluters' consent to face regulation (Tietenberg, 2000).

This traditional political economy may not hold as strongly for climate policymaking. Rents from carbon emissions restrictions will be massively larger than

[2] As pointed out by Cole (2015) and Cole and Grossman (2002b), the view that polluters in an unregulated state have the right to pollute is not correct, because the rights have not yet been legally established. But this reality does not prevent polluters from believing that they have the rights -- or making the argument that they do.

[3] New source performance standards under the Clean Air Act offer an example.

those from traditional regulation, and significantly more visible. This difference is likely to increase the political conflict over rights distributions, with more claimants than the rents available to compensate losses (CBO, 2003). The visibility of rents may also lead to some shifting in the standard paradigm in which environmental rights are grandfathered to polluters. Some recent CO_2 emissions trading programs auction or partially auction permits, with exemptions for privileged polluting sectors declining over time.[4] However, administrative rulemaking of the traditional kind is the way carbon emissions are now being federally regulated in the United States.[5]

It is not just the existence of imperfectly efficient institutional arrangements which distributional issues can explain, but the existence of economically inefficient institutions relative to a status quo without them. Governance institutions in the agricultural sector often have the foundational motive to distribute rents from consumers to farmers in a way which disguises the saliency of the consumer losses and the visibility of the efficiency costs. Repealing the laws and associated administrative apparatus which enable these actions would boost economic efficiency. In the areas of environmental policy and transportation safety, visible political problems, such as the discovery of hazardous waste sites or the crash of an airplane, often lead legislatures to pass binding statutory

[4] Even in the climate policymaking context, however, the prices of carbon are often lower than the social cost of carbon. In such cases, carbon taxes or auctioned tradable permits are again playing the role of user charges. Examples include the Regional Greenhouse Gas Initiative (RGGI) in northeastern United States, the tradable program for CO_2 allowances started in California, and the evolving emissions trading system in the European Union. For details about these programs, see http://www.rggi.org/, http://calcarbondash.org/, and http://ec.europa.eu/clima/policies/ets/index_en.htm.

[5] As is typical in the United States, legal challenges have stalled the implementation of this policy.

obligations with little scope for implementation flexibility. In the United States, the Superfund law and a battery of laws implemented by the Federal Aviation Administration (FAA) are examples. The rulemakings from such statutes do not pass a benefit cost test.

The basic point of this article is to examine a related issue: how conflict over property rights imposes an economic cost which raises the bar for regulatory governance institutions to be economically justified. The case study here is institutions which address environmental problems. To study the normative implications of political conflict over environmental policymaking, we formulate a parsimonious analytical model of a "noisy" political process – one which allows varying degrees of connection between lobbying actions and outcomes – to mimic imperfect political decision-making. The model includes economic parameters for environmental benefits and abatement costs, and contextual, institutional parameters including the relative political power of stakeholders (or under another interpretation of the formulation, the "legal merits" of the policy proposal) and the "technology" of the political process (a parameter which translates lobbying actions into results, holding the level of noise constant). The main policy parameter is the degree to which environmental rights are distributed between polluters and other stakeholders. A contextual parameter is also included which indexes the visibility of rents to stakeholders other than polluters. Pure-strategy Nash equilibria are simulated for the resource costs stakeholders incur in political conflict, and then these costs are incorporated into welfare measures showing the net-value of environmental policy required to justify regulation. The results show that relative political power and the “technology” of lobbying, the visibility of rents, and the degree of noise in the political decision-making will play an important role in the costs of political conflict. Moreover, factors such as perfect information and low noise which improve the efficiency of rights exchanges in private markets can increase the costs of rights exchanges mediated through a political process.[6]

The rest of the article proceeds as follows. Section 2 develops a model of political conflict over environmental policymaking. Section 3 describes Nash equilibria for the resource costs incurred to influence the policy process, and conducts numerical simulation to show how parameter variation affects them. Section 4 incorporates the resource cost measures into value thresholds for environmental governance to be judged to be economically efficient. Section 5 also incorporated uncertain political decision-making into the welfare calculus to compute expected value thresholds for environmental governance to be judged to be economically efficient. Section 5 offers summarizes and offers conclusions.

2. The Model

This section describes a simple model of political conflict which has the fundamental character of the political pressure model introduced by Becker (1983). In this set up, institutional detail is suppressed, and the policy is seen as ultimately arising

[6] As in Libecap (1989), the exchange of well-defined rights is seen here as categorically the same as the clarification or definition of previously undefined rights.

from the weight of interest group pressure placed on a responsive political process.[7] It is necessary to add the qualification that this model implicitly is based on an assumption that interest groups do not have the opportunity to voluntarily negotiate a consensual agreement ex ante. In that case, a two-stage game formulation would be needed with the political contest forming the endogenous disagreement point for a second stage political conflict. We assume that the transactions costs of ex ante bargaining gives the conflictual route as the only option, a fairly realistic assumption given the ubiquity of political-legal conflict over environmental policymaking – at least in the United States.[8]

Given this framework, it is assumed that an environmental authority proposes a policy which a homogenous group of environmentalists and polluters contest. Let B, C, and R respectively denote the exogenous benefits, costs, and inframarginal environmental rents which could be captured by the governance authority if an emissions tax of the conventional type was imposed (or equivalently, if tradable permits were auctioned). That is, "R" is the maximum rent which could captured from the polluter by

[7] Take the example alluded to in the previous section about "regulator" focusing more on the level of regulation than on the allocation of rents. This predilection is ultimately shaped by the enabling legislation, which is subject to interest group pressure. The Becker modeling approach implicitly compresses into one stage the legislative and regulatory stages, as they are differentiated into the United States. Thus, political pressures around the legislative enactment *ex ante*, and legal actions around the policy in the regulatory implementation stage *ex post*, are aggregated into "combined" pressure which influence whether the policy is ultimately implemented, and the combined resource costs of pro and con pressures over the policymaking life cycle. See Krutilla and Alexeev (2012, 2015) for recent applications.

[8] This context differs from that of civil litigation where an initial "out-of-court" bargaining stage is common, and only those cases which cannot be consensually resolved rise to the level of an adjudicated legal contest (Cooter and Rubinfeld, 1989). The analogue in the environmental policy area would be the negotiation of "voluntary" arrangements -- among environmentalists and polluters, or polluters and governments, or self-regulation by firms -- which substitute for statutory restrictions. In the United States at least, such voluntary measures account for a relatively small share of pollution control.

the governance institution for a given policy proposed. It is assumed that $B > 0,\ C > 0$, and $R \geq 0$. The magnitude of B and R will be varied in sensitivity analyses while holding C constant. This implies that the level of pollution control does not change in the simulations. The parameter B can be varied for an absolute level of pollution control due to possible difference in environmental damages.[9] Varying R for a given level of pollution control requires some not-fundamental technical conditions which are assumed for convenience. To see what they are, let e be the unregulated emissions level, σ the share of emissions remaining *ex post* after regulation, so that σe and $(1-\sigma)e$ comprise the division between uncontrolled and controlled emissions. Let τ be the marginal rent associated with the last unit of uncontrolled emissions, and assume that marginal abatement costs can be linearly approximated as rising from 0 to τ as the level of emissions control goes from zero to $(1-\sigma)e$. On these assumptions, $R = \sigma e \tau$ while $C = .5(1-\sigma)e\tau$, giving $\dfrac{R}{C} = \dfrac{2\sigma}{(1-\sigma)}$. Suppose that $\sigma = .9$, implying pollution has been reduced by 10%. This implies $\dfrac{R}{C} = \dfrac{2 * .9}{(.1)} = 18$. Now suppose the goal is to evaluate $\dfrac{R}{C}$ for a 20% level of emissions control, corresponding to $\dfrac{R}{C} = \dfrac{2 * .8}{(.2)} = 8.$ Maintaining the absolute level of pollution control across this comparison necessitates achieving the 20% reduction by changing the size of the firm. Without loss of generality, assume the firm

[9] Specifically, we assume that the variation of B for a given C results from differently-slopped marginal benefit curves, all of which intersect the marginal cost curve at a point which gives "C" as the abatement cost. That is, the marginal conditions are held constant in sensitivity analyses.

size in the first instance is $e = 1$, so that the 10% reduction corresponded to an absolute pollution reduction of $.1$ Solving for the size of the firm corresponding to a 20% reduction which gives the same absolute pollution reduction as in the first instance we have: $.1 = .2e \rightarrow e = .5$. That is, the firm size in the second case is ½ the size of the first. Changing firm size to give different $\frac{R}{C}$ ratios for a given level of pollution control is implicitly the method we are using for varying $\frac{R}{C}$ in the simulations conducted in the article.

Lastly, it is also assumed that the demand for polluter's output is infinitely elastic, implying that if R is captured by the governance authority, the impact is fully borne by the polluter. Again this assumption is a technical, rather than fundamental, assumption in our context.[10]

There are two additional parameters in the model. The first is $\delta \in [0,1]$, a policy parameter for the fraction of the potentially capturable inframarginal rents (R) actually specified in the policy proposal to be transferred from the polluter to the governance institution. For δ to be varied without disrupting the marginal conditions which are implicitly establishing C, B, and R, the policy must be assumed to be implemented as a

[10] On the standard assumption in the rent seeking literature that the incentive to rent seek over supernormal returns is the same as to avoid absolute losses, the incidence of R does not matter to political behavior. The only reason we make the assumption that the firm bears this cost is to maintain consistency with other assumptions. For example, if the incidence of pollution control was partially shifted to consumers, there could be market responses which would feedback into the regulated market, affecting the level of pollution control and abatement costs. This effect would be inconsistent with the assumption that the level of pollution control and C does not change as R is varied, so we rule out the possibility.

tax-refund system (see Alexeev et al., 2016; Farrow, 1995,1999; Krutilla and Alexeev, 2015; and Pezzey, 1992, 2003). This method of taxation can be conceptualized in two ways. First, as a Pigouvian tax which exempts an initial range of inframarginal rents up to a threshold, as described in Pezzey and Jotzo, 2015. This threshold can be varied from zero, giving the conventional emissions tax, to “total”, which gives the rights-sharing equivalent of a conventional regulatory standard or grandfathered tradable permits. The equivalent-rights sharing can also be achieved using a tax-refund system in which conventional emissions tax is imposed, but the collected rents are rebated back to polluters’ lump sum to a degree corresponding to the rents generated from the emissions exemption thresholds just described. In fact, such a system has been implemented to reduce NOx emissions for stationary source emitters in Sweden (Sterner and Isakkson, 2006).

We also include a “visibility” parameter, $\nu \in [0,1]$, which reflects the degree to which environmental beneficiaries perceive the infra-marginal rents. The limiting case $\nu = 0$ indexes the situation where environmental beneficiaries do not perceive the rents, while $\nu = 1$ is the case that beneficiaries have the same perception of the rents as polluters. A case in between these two extremes, say at $\nu = .5$ .would suggest that the perception of the rents is diffuse enough that the payoff of seeking the rent is only one half of the rents available.

Given this framework, it is assumed that polluters lobby against the policy to avoid their losses – the sum of abatement costs plus environmental rentw captured by the environmental authority, $(C + \delta R)$, while environmentalists lobby for their perceived gains, $(C + \nu\delta R)$. As in Becker (1983), this process is modeled as a one-shot simultaneous move game, giving in our case:

$$\max_{C_1} \xi_1 = (B + \nu\delta R)\rho\left(C_1, C_2\right) - C_1, \tag{1}$$

$$\min_{C_2} \xi_2 = \left(C + \delta R\right)\rho\left(C_1, C_2\right) + C_2, \tag{2}$$

The variables C_1 and C_2 are the economic cost of lobbying effort by environmentalists and polluters respectively, while ξ_1 and ξ_2 are the corresponding expected net-pay offs. It is assumed that $B > C$, implying that only economically efficient proposals are proposed – in the conventional sense of the term "economically efficient." Thus, the conventional normative ideal would be for the proposed policy to pass the political test.

To get some intuition about the model described in (1) and (2), notice that for a conventional regulatory standard, $\delta = 0$, so that the rent terms (R) drop out of the equations. In this case, environmentalists simply lobby to receive the benefits, B, while polluters just lobby to avoid the pollution abatement costs, C. On the other hand, if an emissions tax of the conventional type is imposed, $\delta = 1$, and the polluter now lobbies to avoid both the pollution abatement costs C, and the loss of environmental rents, R. The behavior of environmentalists in this situation depends on the visibility of the rents. If the rents are not visible, $\nu = 0$, environmentalists lobby only to receive B, as before. This is the classical political economy described by Buchanan and Tullock (1975). On the other hand, if the rents are visible, $\nu = 1$, then beneficiaries lobby over R as well as B, giving (B+R) as the payoff. In this situation, a zero-sum rent seeking contest is embedded within the political contest. As noted, this situation may be closer to the political economy surrounding carbon emissions control, as discussed in CBO (2003).

Equations (1) and (2) also include the political influence function, $\rho(C_1, C_2)$, which represents the probability of the environmental policy's acceptance as a function of lobbying resource costs.[11] An adaptation of the contest success function described in Tullock (1980) is used to represent this functional form:

$$\rho(C_1, C_2) = \frac{C_1^{\ r} + \lambda}{C_1^{\ r} + \alpha C_2^{\ r} + 2\lambda} \tag{3}$$

The α parameter allows for different kinds of asymmetries in the relative effectiveness of lobbying effort. When $\alpha < 1$, a unit of lobbying effort by the beneficiaries has a higher relative impact on the marginal probability of the policy's acceptance than a unit of lobbying effort by the polluter. With $\alpha > 1$, polluter lobbying has the higher effect on the marginal probability. It is assumed that $\alpha \in (0, \infty)$. Asymmetries in the marginal effectiveness of lobbying effort could arise from difference in the political power of the competing groups, equity norms, or other kinds of biases which would predispose policymakers toward one outcome or the other, e.g., difference in the degree to which the policy proposal is consistent with the objectives of the legal or other authority which motivates it.

[11] Because the outcome is a binary go no-go decision about the policy proposal, "$\rho(C_1, C_2)$" is interpreted as the probability of political success, rather than as a share of rents distributed to claimants. The latter is a common interpretation of rent-seeking contests with risk-neutral claimants.

The r parameter represents the returns to lobbying effort, often described as the "technology of rent-seeking" (Hillman 2014). This parameter has been shown in the rent-seeking literature to have a crucial effect on the degree to which rent-seeking actions dissipate rents. It is assumed that $r \in (0, \infty)$. However, we will end up restricting the domain of r to give pure-strategy solutions. This restriction always caps the upper limit of r to between 1 and 2 in standard rent-seeking contests (Baye et al., 1994; Perez-Castrillo and Verdier, 1992)

The parameter λ is a "noise" parameter, as described in Dasgupta and Nti (1998) and Amigashi (2006). When, $\lambda = 0,$ lobbying outcomes are defined by the standard Nash equilibria. As λ increases, political outcomes become less sensitive to the value of the resources invested in lobbying, and solutions depart from the standard Nash solutions. In the limit as $\lambda \to \infty$, $\rho \to .5$ regardless of the resources invested in lobbying. Note that if $\alpha = r = 1, \lambda = 0,$ and $C_1 = C_2$, $\rho \to .5$. In short, parameter variation leads to asymmetries in the probability of political outcomes per unit of resources invested in lobbying.

3. Model Solutions

First order conditions for the full model give transcendental equations which cannot be simplified to give analytical solutions. Analytical solutions can be derived when either the r or the λ parameter is fixed. Letting $r = 1$ gives:

$$c_1^* = \frac{\left(\phi\delta\nu + \beta\right)^2 \left(1 + \phi\delta\right)\alpha}{\left(\alpha\left(\phi\delta + 1\right) + \phi\delta\nu + \beta\right)^2} - \lambda \qquad (4)$$

$$c_2^* = \frac{\left(\phi\delta + 1\right)^2 \left(\phi\delta\nu + \beta\right)\alpha}{\left(\alpha\left(\phi\delta + 1\right) + \phi\delta\nu + \beta\right)^2} - \frac{\lambda}{\alpha} \tag{5}$$

while setting $\lambda = 0$ gives:

$$c_1^* = \frac{\alpha r \left(\delta\nu\phi + \beta\right)^{r+1} \left(\delta\phi + 1\right)^r}{\left(\alpha\left(\delta\phi + 1\right)^r + \left(\delta\nu\phi + \beta\right)^r\right)^2} \tag{6}$$

$$c_2^* = \frac{\alpha r \left(\delta\nu\phi + \beta\right)^{r} \left(\delta\phi + 1\right)^{r+1}}{\left(\alpha\left(\delta\phi + 1\right)^r + \left(\delta\nu\phi + \beta\right)^r\right)^2} \tag{7}$$

The new variables reflect a normalization using the C parameter. Specifically, $c_i^* \equiv C_i / C$ indicates how resource costs devoted to political conflict (C_i) compare to conventionally-measured economic costs (C), with $i = (1,2)$ denoting environmentalists and the polluters respectively; $\beta \equiv B / C$ is the policy's conventionally-measured benefit-cost ratio; and $\phi = \frac{R}{C}$ is as defined before.

It is useful to define $\theta \equiv \sum_i c_i$, the total lobbying costs incurred by both agents in relation to the policy's abatement costs. As might be surmised from the functional forms displayed in (4)-(7) only $\frac{\partial\theta}{\partial\lambda}$ can be signed. The derivatives from (4) and (5) show that politically-induced resource costs decrease as decision-making noise increases. Modifying (4)-(7) by letting $\alpha = r = 1$, the signs for $\frac{\partial\theta}{\partial\gamma}$, with $\gamma \in (\beta, \delta, \nu, \phi)$, will all be positive. That is, the total resource costs devoted to policy conflict are increasing in the benefit-cost ratio, the share of rents captured by the environmental governance authority, the visibility of the rents to the beneficiaries, and the magnitude of the rents in relation to abatement costs.

Numerical simulations are conducted to get a sense of the magnitude of these directional effects, and also to show the effects of varying the α and r parameter. Second order conditions and positive profit conditions consistent with pure strategies are satisfied in the numerical simulations for the parameter conditions assumed, except where specifically indicated.

Figure 1 plots θ for different values of the δ, ν, α and r parameters in one-way sensitivity analyses; $\phi \equiv R / C$ is indicated on the horizontal axis running from 0 to 25. As noted before, $\phi = \dfrac{R}{C} = \dfrac{2\sigma}{(1-\sigma)}$, with σ and $(1-\sigma)$ respectively the share of the unregulated and regulated emissions *ex post*. Thus, $\phi = 25$ corresponds to a policy which reduces pollution by a relatively small amount $(\approx 7.4\%)$, resulting in a relatively large emissions base for the inframarginal rents $(\approx 92.6\%)$. On the other hand, $\phi = 0$ corresponds to a total pollution restriction which gives no inframarginal rents. In all of the simulations, $\beta = 3$ and $\lambda = 0$, while the base settings for the other parameters is $\alpha = r = \nu = \delta = 1$ when they are not being explicitly varied.

Panel A shows how θ responds to varying degrees of rent capture by the governance authority. When no rent is captured ($\delta = 0$ lowest line in the figure), differences in the environmental rents revealed by the policy (ϕ varying between 0 and 25) do not have a significant impact on θ . However, at lower levels of regulation/larger firm size for which significant environmental rent is revealed $(\phi \rightarrow 25)$, difference in the degree to which the government captures (δ varying from 0 and 1) normatively matters. At $\phi = 25,$ a policy which has the property right structure of a standard regulation ($\delta = 0$) generates resource costs over the policy conflict equal to .75 the policy's abatement cost. For a policy with the property rights equivalent of a conventional emissions tax or auctioned tradable permits $(\delta = 1)$, resource costs over the policy conflict are about 13.6 times higher than the policy's abatement costs.

Panel B of Figure 1 shows the impact of varying the parameter for the visibility of the environmental rents to the environmental beneficiaries (Again, the rents are always visible to the polluters, who will lose the rents in a zero-sum transfer if the governance authority collects them). When rents are not visible to the environmental beneficiaries (the $\nu = 0$ bottom line), ϕ will increase from about .75 to about 2 as ϕ varies from zero to 25. That is, the resource costs polluters devote to political conflict increase as the amount of rent increases, given the base parameter setting in Panel B in which the government policy is to collect the rents $(\delta = 1)$. Starting from the point where $\nu = 0$ and $\phi = 25$, and increasing the visibility parameter from $\nu = 0$ to $\nu = 1$, theta will increase from around 2 to about 13.6. The $\nu = 1$ case would be the expectation for the resource costs incurred in relation to abatement costs for auctioned permit programs or emissions taxation for a pollutant like carbon emissions for which emissions are likely to be visible, and for the case where the degree of emissions control is not very large (again around 7.4%). The $\nu = 0$ case could be the expectation for θ for a local pollutant like SO_x for which the rents are not visible except to the regulated industry. Because there is a non-linear relationship between ϕ and the magnitude of emissions reductions, a 50% cut in emissions will result in a ϕ of about 2. Considering the impact of collecting rents for carbon taxation versus that of a local pollutant at this point, it can be seen that variation in the visibility of rents at $\phi = 2$ is almost not significant. Thus it is the range of emissions reductions up to 50% that the visibility of rents will normatively matter. Of course this statement only necessarily holds for the parameterization shown.

We now turn to the impact of the relative political power parameter, α. Panel C shows that θ is initially rising in α, but then declines. The easiest way to understand this pattern is to consider the consequences when α is at the limiting extremes of $\alpha = 0$ (environmentalist has total political power) and $\alpha = \infty$ (polluter has total political power). In either case, the contest ends, with the party having the dominant power position prevailing without the need to devote resources to lobbying. It must be the case then that starting from the point where $\alpha = 0$ and increasing α raises the marginal payoff to the polluter of contesting the rights, and brings them into the contest. That increases θ. But as α increases far enough beyond the point where the two parties have symmetric political power $(\alpha = 1)$, the marginal value to the beneficiary of contesting the rights eventually diminishes, and θ declines again (again reaching zero in the limit as $\alpha \rightarrow \infty$)

The final parameter considered is "r", shown in Panel D. As in the standard rent-seeking contest (Baye et al., 1994; Perez-Castrillo and Verdier, 1992; Van Long, 2013), pure strategies are observed for $r \leq 1$. However, for $r = 1.4$ and $r = 1.6$ lines, there is some initial range for $\phi < 5$ where pure strategies do not exist (indicated in Panel D by horizontal lines along the axis in this range). In general, θ is rising in r for the parameterization shown, and θ rises to very high levels for $r > 1$ when the inframarginal rents are substantial. At $\phi = 25$, for example, θ rises from 13.6 at $r = 1$ to 15.4, 17.3, and 21.9 as r increases respectively to 1.2, 1.4, and 1.6. (Pure strategies do not exist for $r > 2$) On the other hand, when $r = .5$, $\theta = 6.2$ – about 28% of the value of θ at $r = 1.6$. In sum, θ is very sensitive to variation in the r parameter. This outcome parallels the result observed in the rent-seeking literature about the impact of the r parameter on the degree of rent dissipation.

4. Value Thresholds for Efficient Environmental Governance

We now turn to the implications of the previous results for establishing the margin of economic activity which can be efficiently transferred from the private to the public sector. As a first approximation, the metric for demarcating the dividing line between economically-efficient private and public governance is given by a welfare standard defined as follows:

$$W = (B - C) - \sum_{i=1}^{2} Ci^* \tag{8}$$

"W" requires that the conventionally measured net-benefits, $(B - C)$, cover the total resource costs of the political conflict, $\sum_{i=1}^{2} Ci^*$, associated with formally defining the environmental rights and transitioning management responsibility. For $W > 0,$ it is economically efficient to bring economic activity within the sphere of public governance, notwithstanding the transactions costs of organizing a "new contract" over the property rights – using terminology from Libecap (1989). For $W < 0,$ the benefits of the rights definition are not worth the costs of the political conflict, and it is more economically efficient to leave the rights in the undefined state of the *status quo ante.*

Using previous notation (8) can be re-expressed to give:

$$\beta = 1 + \theta(\delta, \nu, \alpha, r, \beta, \phi, \lambda) \tag{9}$$

Notice than when the costs of the political conflict are zero – $\theta(\bullet) = 0$ – equation (9) gives the standard transaction cost-free benefit-cost ratio. Solving out for β on the right hand side gives:

$$\beta^* = \theta(\delta, \nu, \alpha, r, \phi, \lambda) \tag{10}$$

The measure β^* is a "break-even" ratio that defines the boundary between economically efficient and inefficient public governance. Equation (10) does not have an analytical solution, so numerical simulations are used to plot β^* for the same parameter variations as for θ illustrated in the previous section (See Figure 2). Following directly from the previous results described in Figure 1, β^* is rising with respect to all parameters holding the other parameters constant at their indicated values, and parameter variation matters more for $\phi > 2$ (emissions reductions less than 50%) than for $\phi < 2$ (emissions reductions greater than 50%). For a total level of emissions control (i.e., $\phi = 0$) break-even ratios are in the neighborhood of $\beta^* = 2$. At $\phi = 25,$ the maximum β^* values across the parameter configurations range from over 17 to around 23. This is well beyond the range of the benefit-cost ratios found in RIAs conducted of significant environmental regulations in the United States. However, actual regulations in practice impose a mix of control levels, and most significantly, $\delta = 0$ in virtually all of them. The break-even ratios in this case (for the default parameter settings simulated) are closer to $\beta^* = 2$, independently of the infra-marginal rents generated.

5. Expected Value Thresholds for Efficient Environmental Governance

The welfare metric just described gives a liberal estimate of the size of the economically efficient space for public governance, in the sense that it does not consider the welfare consequences of the uncertainty inherent in political decision-making. Illuminating demonstrations of such uncertainty are offered by the two most notable attempts in the United States to reduce energy consumption and carbon emissions. The first was the proposal by the Clinton administration in 1993 to impose a broad-based energy tax. This proposal was fiercely opposed in Congress and ultimately ended up as a small sales tax on electricity which was repealed in 1998. The second attempt is the cap-and-trade carbon emissions trading program proposed by the Obama administration in 2009. This plan was never passed in Congress. In short, there was ultimately no net-benefit from either of these legislative proposals, yet resource costs were incurred in the political conflicts around them.

This result might seem normatively similar to outcome of a classic rent-seeking contest, but there are significant differences. Specifically, policymaking is exogenous in the rent-seeking literature, and the focus is exclusively on the degree to which policy-associated rents are dissipated. As such, rent-seeking contests are always zero sum. This context differs from that just described, and modeled in this paper, in the sense that policymaking is endogenous, and a rent-seeking contest is embedded within the policy conflict, so that the contest is never zero sum. In fact, this is the standard form for the property rights negotiation in environmental policymaking. Even when inframarginal rents are zero, there will be political costs incurred by stakeholder over the policy's benefit-cost structure. This structural detail makes it normatively matter whether or not policies pass the political test. If policies are economically efficient in the conventional sense, $\beta > 0$, there is some chance that the conventional net-benefits can cover the associated political costs. The relevant question to ask *ex ante* is whether the expected net value of the policy, $\rho(B - C)$ is greater than the expected resource costs of the property rights conflict, $\sum_{i=1}^{2} Ci^*$, in view of the uncertainties inherent in political decision-making.

Using the notation previously introduced and adopting this welfare metric gives the following:

$$\beta = 1 + \frac{\theta(\delta, \nu, \alpha, r, \beta, \phi, \lambda)}{\rho(\delta, \nu, \alpha, r, \beta, \phi, \lambda)} \tag{11}$$

This expression is the same as (9) except that the probability term shows up in the denominator. Since $\rho < 1$, adding decision-making uncertainty raises the value threshold for efficient public governance. Solving out for β gives the same structure for the reduced form as in (10).

Again we follow the strategy of using numerical simulation to indicate thresholds as a function of parameter variation. Figure 3 shows that the general pattern of results observed in the previous section is repeated for the expected value thresholds, with the exception of the α and ν parameters. β^* is strictly increasing with α in Panel C, unlike previous case shown in Figure 2 where β^* initially rises and then declines with increasing α values. From equation (3), it can be seen that $\frac{\partial\rho}{\partial\alpha} < 0$; that is, the probability that the policy passes declines with increasing relative political power of the policy's opposition. This reduces the expected value of the policy as α increases, raising value thresholds required for expected value to be positive. The other notable distinction is the lack of differentiation of β^* values for different levels of ν. At the present stage of research, we do not yet have an explanation for this result.

Notice that the magnitudes of the values for β^* in Figure 3 are significantly larger than those in Figure 2 – for equivalent parameter values. Holding α and r at 1, the maximum value for Panel A in Figure 3 is 27.5 compared with about17 in Figure 2; for *r*, the comparison is 26.5 to about 17. The maximum value for β^* in Panel D in Figure 3 is 29 compared 23 in Figure 2. The value for $\alpha = 2$ in Panel C of Figure 3 is close to 60 compared with about 16 in Panel C of Figure 2. These differences are at the upper end of the range for ϕ. In the other limiting extreme where $\phi = 0,$ the expected value thresholds with political uncertainty are in the neighborhood of $\beta^* = 3$ for Figure 3 compared with around $\beta^* = 2$ in Figure 2.

5. Discussion and Conclusion

In the classic Coase literature, bargaining transactions costs define the boundary for economically efficient divisions in environment management between the private and public sectors. When bargaining transaction costs are prohibitively high, it may be economically efficient for the government sector to take over management responsibilities. The environmental economics literature suggests a standard benefit-cost test to determine the efficiency of public governance in this situation.

Although the standard Coase and environmental economics literature are often seen as offering polar perspectives, both share a common premise: that the rights assignment is exogenous. In an endogenous rights setting, Libecap (1989) documents the impact of distributional conflicts on the emergence of organized governance regimes. This article uses a highly stylized model to measure the welfare costs of this category of property rights conflict, and assesses the implications for the size of the economically efficient space for public environmental governance.

The general model does not have analytical solutions; consequently, numerical simulations are used to assess the effects of parameter variation. The parameters considered include structural characteristics of the policy – the policy's environmental benefits and costs – and structural characteristics of the political decision-making, including the noise in process and the responsiveness of political decision-making to the resources devoted to influencing it -- holding the level of noise (and other parameters) constant. Also included is a parameter for the degree to which the beneficiaries of the policy perceive the policy-associated rents. Finally, a policy parameter is included which indicates the degree to which environmental rents are transferred from polluters to the government.

Three measures are used to assess the welfare costs of establishing environmental rights. First, the ratio of the resource costs associated with political conflict to the policy's abatement costs. Depending on the parameter variation, this ratio can vary from less than 1 to over 20. In short, the costs of transaction cost of political conflict can be two orders of magnitude larger than the policy's abatement cost. It is an interesting reality that this welfare cost is universally ignored in standard policy analyses of environmental policymaking. However, if the benefits of the policymaking are large enough to cover the abatement and political transactions costs, moving economic activity from the private to the public sectors will still be economically efficient. The second welfare measure provide a break-even standard to make this assessment. Again it exhibits a wide variation. Finally, a third standard is used which incorporates the uncertainty of the decision-making outcome into the break-even welfare standard. This standard requires that the expected net-benefits of the policy cover the resource costs over property rights conflict. Breakeven benefit-cost ratios by this measure can vary between 25 and 50 for some parameter configurations before environmental governance by the public sector becomes economically efficient.

The allocation of property rights between polluters and other stakeholders is the one policy parameter which can be used to alter political transaction costs. In fact, the disposition of environmental rents is the essential property rights conflict in environmental policymaking, and is becoming increasingly policy-relevant in era in which climate policymaking is the foremost environmental challenge of the time. For the limited parameter variation assessed, higher economic costs are associated with policies which transfer rents from polluters.

Political bargains are often observed in practice in which more stringent environmental regulation is negotiated in exchange for private rent capture. Most tradable permit systems in the United States have grandfathered environmental rights in exchange for more stringent compliance schedules. Examples include the Acid Rain Program established under the 1990 Clean Air Act Amendments, and the RECLAIM trading system established by the South Coast Air Quality Management District in California (Tietenberg 2000). Pezzey and Jotzo (2013) suggest the same kind of tradeoff for climate policymaking. Specifically, they suggest that higher carbon prices will only be politically feasible if thresholds are established for the exemption of infra-marginal emissions. The preliminary results of this research suggest that such rent sharing can expand the space for economically efficient environmental management in the public sector.

Figure 1. The Ratio of Resource Costs in Political Conflict to Pollution Abatement Costs

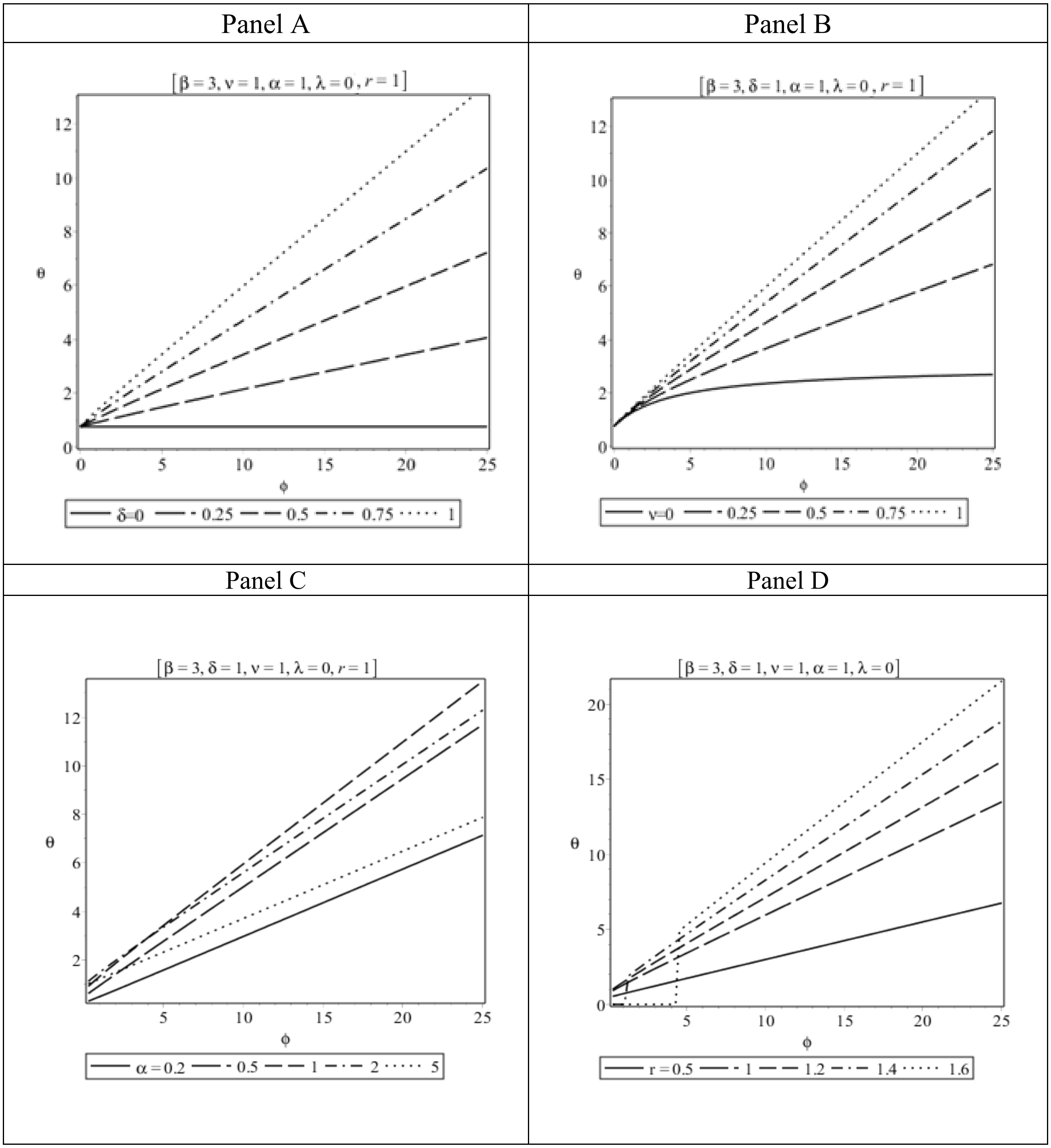


Figure 2. Value Thresholds for Economically Efficient Environmental Governance

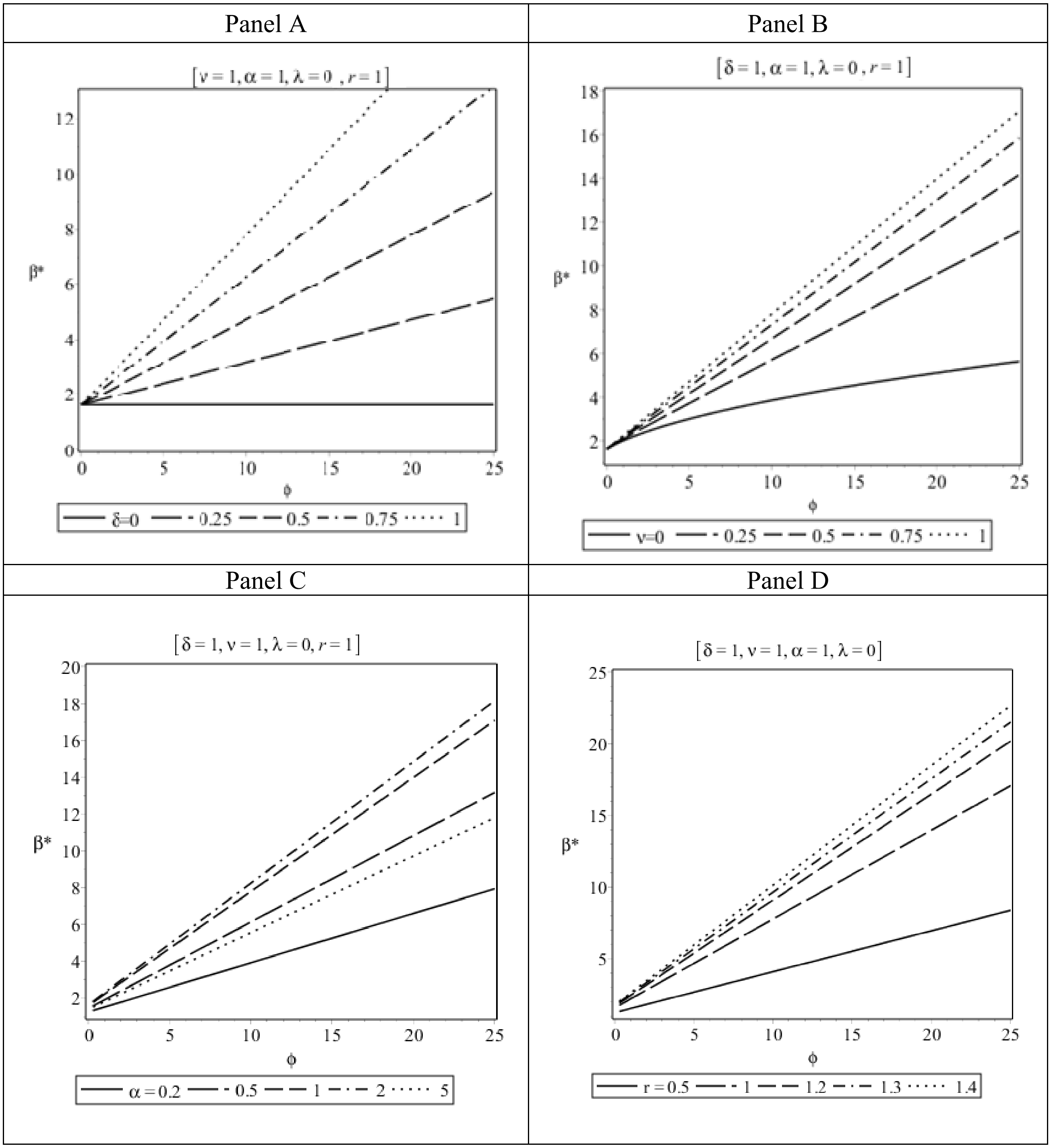
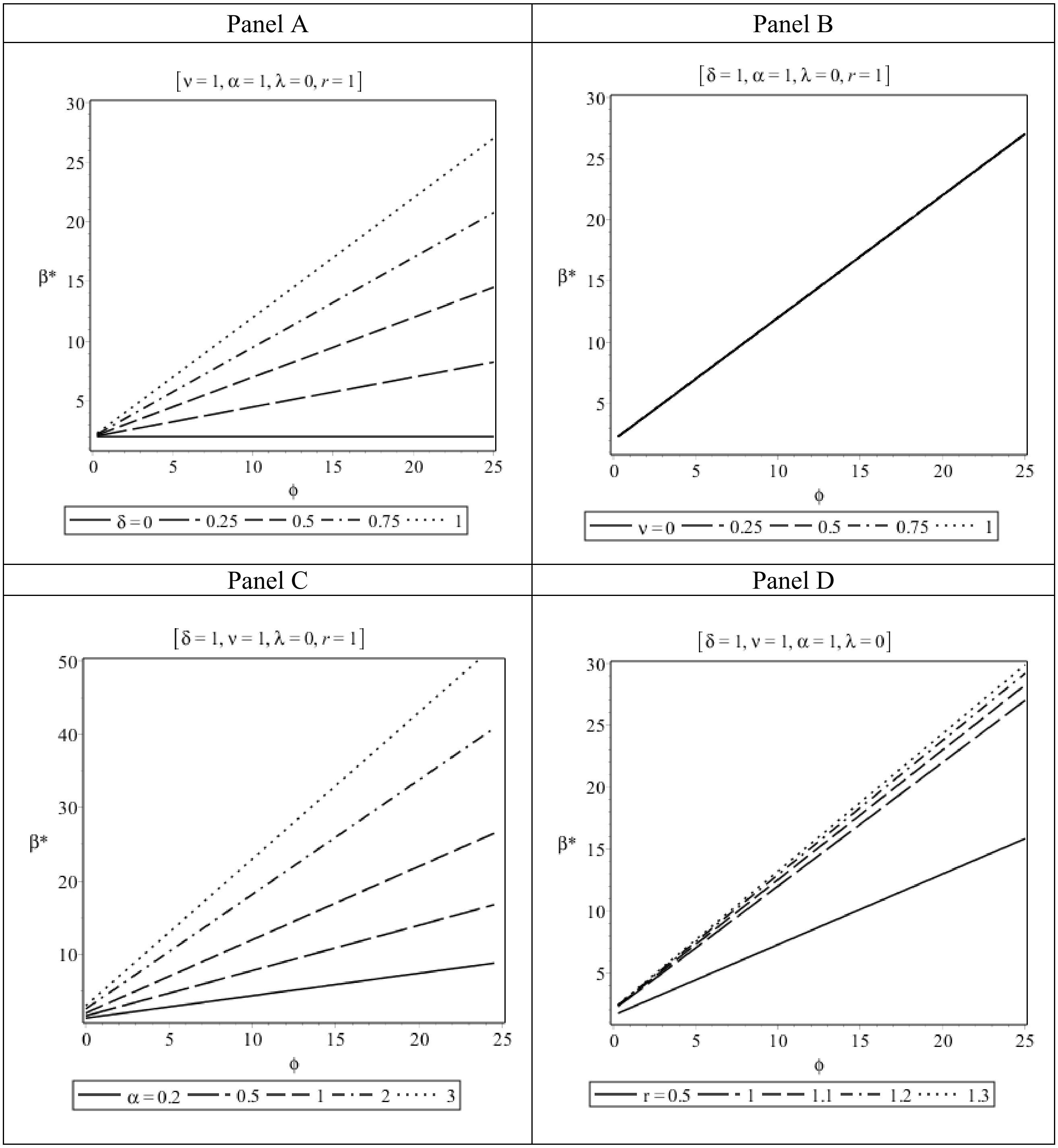


Figure 3. Value Thresholds for Economically Efficient Environmental Governance with Uncertain Political Decision-Making



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