The Use of Group-Level Approaches to Environmental and Natural Resource Policy

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Introduction

Environmental economists have a long history of studying alternative policy instruments aimed at improving environmental and natural resource management. In most cases, the particular policies under study target individual polluters or resource users. Examples of such policies include limits on a firm's emissions, taxes on a firm's polluting inputs or outputs, and information disclosure about a firm's production practices or emissions. This article discusses policies that are instead applied at the group level. The defining feature of a group-level policy is that rewards or penalties are based on group performance rather than the practices or performance of individuals, or rights are allocated to a group rather than individuals (Kotchen and Segerson 2018). For example, rather than each firm facing a tax that depends only on its own emissions, each firm could face a tax that is based on ambient levels of pollution (i.e., pollution levels or concentrations measured in a given waterbody or airshed), which depend on the combined emissions from all contributing sources (Xepapadeas 2011). Similarly, rather than paying individual landowners for their conservation efforts, a payment-for-ecosystem-services (PES) program could make payments to a community or village based on its collective conservation activities or outcomes (Kerr, Vardhan, and Jindal 2014). In addition, an entire industry could be threatened with costly regulation if it fails to collectively meet pollution control objectives through some form of self-regulation or voluntary pollution control (Segerson 2017). Finally, rather than allocating fishing rights or limits on allowable incidental catch of other species (bycatch) to individual vessels, the rights could be allocated to a fishing cooperative or an entire fleet (Holland 2018).[1]

Although there is a large literature on collective (group) action to improve resource management (e.g., Ostrom 1990, 2000), the literature on regulatory policies that are based on group-level rewards, punishments, or collective rights has been much more limited. As we

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will discuss, there are a number of studies that focus on the use of specific group-level policies in particular environmental and natural resource contexts. What is missing is a synthesis across different contexts that can provide general insights about when and how regulators might use group-level polices as an effective and efficient regulatory approach. This article provides a first step toward providing such a synthesis.

We are interested in how group-level approaches can, do, and should fit into the proverbial "toolbox" of policy instruments for environmental and natural resource management. We aim to identify theoretical and empirical insights and lessons learned about the design and implementation of these instruments by examining group policies in different contexts. Rather than provide a comprehensive review, we focus on the use of group-level policies in four specific contexts: agricultural pollution from diffuse sources (i.e., "nonpoint" pollution[2]), PES, industry-wide voluntary programs, and fisheries. We also briefly discuss the use and performance of group-level policies at the national level under the U.S. Clean Air Act (CAA) and Clean Water Act (CWA). An important conclusion that emerges from examining these various contexts is that group-level policies are most likely to be effective when rewards and/or penalties are designed to provide strong incentivizes for groups to meet targets in a cost-effective way. We also find that to the extent that this requires coordination within the group, it is important that the group has or can create its own institutions or mechanisms to facilitate and enforce that coordination.

The remainder of this article is organized as follows. The next section presents a conceptual overview of group-level policies, highlighting general issues that arise in the design and use of these policies.[3] We then examine their application in the four specific contexts listed above. For each context, we discuss how the most relevant general issues apply, provide examples of group policies that have been implemented, and summarize the evidence regarding their impact. We then briefly discuss the parallels between these contexts and the group performance provisions of the U.S. CAA and CWA. We conclude with a summary of insights and lessons learned about the design and implementation of group-level policies.

A Conceptual Overview of Group-Level Policies

Broadly speaking, a group is any collection of individuals (e.g., landowners) or organizations (e.g., firms). There are a number of different ways in which groups might form or be defined. A group could represent a natural or feasible management unit, such as all firms within a biophysical unit (e.g., a watershed), a political jurisdiction (e.g., a state or municipality), an economic sector/industry, or some combination of these. Alternatively, a set of individuals or firms might voluntarily choose to form a group, such as a group of owners of fishing vessels who form a cooperative. The definition of the relevant group might be part of the policy design (if a regulator can choose which firms to include in the group), or it might be determined by other factors (such as legal or social relationships among group members).

[2]The term "nonpoint" pollution is used to distinguish between pollution that stems from diffuse sources (such as agricultural land) and pollution that stems from more concentrated or "point" sources (such as smokestacks or discharge pipes).

[3]For a more formal analysis of a unified model of group-level policies, see Kotchen and Segerson (2018).

Our primary interest here is in the design of policies aimed at correcting externalities (i.e., reducing the costs) that members of the group collectively impose on others outside the group.[4] For example, agricultural pollution from a group of farmers imposes costs on downstream water users, and reduced ambient air quality resulting from emissions by a group of firms in a given area or sector can impose health costs on neighboring residents. Likewise, fishing by vessels in a given fleet can impose ecological damage on marine environments and lead to incidental bycatch of threatened marine mammals valued by society as a whole. In the absence of policy, when group members are making their production decisions, they will generally have little (if any) incentive to consider the costs they impose on others outside the group. It is the existence of such externalities that typically provides the rationale for some policy intervention.

There are at least four reasons why a policy intervention might target the group as a whole rather than individuals within the group:

1. In some cases, individual actions may be very difficult (or even impossible) to monitor, while a group outcome is more easily monitored. For example, a regulator might find it difficult or prohibitively costly to monitor the activities of individual farmers polluting a given waterbody, but easy to monitor the resulting water quality (or ambient concentrations of pollutants) in that waterbody.
2. Even when individual activities can be monitored, group-level policies may have advantages when greater coordination among group members is needed to improve efficiency. For example, in the case of fisheries, coordination among vessels within a fleet might help spread fishing activity, thereby reducing congestion, and sharing information about the location of threatened or endangered marine mammals or other fish species can help avoid incidental bycatch.
3. Group-level policies might be the only sensible approach when resources are owned jointly by a group. For example, in many developing countries, land is collectively owned or managed by a community or village, making the community or village the relevant level for applying policies.
4. Even when resources are individually owned, transaction costs could be lower under a group approach. For example, when seeking to reduce deforestation, contracting with a single entity that represents a group may be less costly than contracting with many small individual landowners. This is similar to (1) if monitoring costs are viewed as a form of transaction costs.

Regardless of the rationale for using a group approach, imposing penalties or rewards based on group performance or allocating rights to groups creates an interdependency among the group members. This interdependency is above and beyond any interdependence that might

[4]In some cases, members of the group may impose externalities on each other as well. Within-group externalities can stem, for example, from spatial interactions such as congestion or use of a common-pool resource (where additional extraction or harvest by one member of the group reduces the amount available for other members of the group). In such cases, the group itself has an incentive to self-organize to address the within-group externality. As noted earlier, there is a large literature that examines the conditions under which self-organization to address over-exploitation of such resources is likely to occur and be effective (e.g., Ostrom 1990, 1994). A thorough review of this literature is outside the scope of this article, although many of the insights discussed below have roots in it.

arise for other, non-policy-related reasons (e.g., from use of a common pool). Specifically, when the aggregate measure of the group's performance or activity (e.g., aggregate emissions, ambient pollution concentrations, combined resource use or degradation) determines the rewards or penalties faced by individuals, each member's own costs, benefits, or both depend not only on the member's own actions but also on the actions of all other group members. Thus, introduction of a group-level policy will create a policy-induced interdependency among firms or individuals within the group, which is a critical feature of all group policies.

This policy-induced interdependence creates two potential issues: (a) incentives to free-ride and (b) incentives to coordinate, or even collude.

Incentives for Free-Riding

Free-riding may occur if some firms are able to benefit from an improvement in group performance even if they do not contribute to the cost of ensuring that improvement. Importantly, when it comes to meeting group-level targets, free-riding can take two different forms: (a) all firms "undersupply" and hence the environmental objective is not met[5] and (b) some firms undersupply and others oversupply. In the second case, the environmental objective can still be met (i.e., there is no "under-provision" at the aggregate level), but the under-provision by some members means that the objective is not met at least cost.

There are several ways in which policy design can address the potential for free-riding. One option is to appropriately structure rewards and penalties (Kotchen and Segerson 2018). For example, even if individual actions cannot be observed, policy parameters (tax or subsidy rates, thresholds, etc.) can be set so they still induce efficient decisions by all group members.[6] This simply requires that rewards and/or penalties for meeting or exceeding group targets are structured such that each group member faces the full marginal social cost and benefit of their actions. For example, under a policy that taxes individuals within a group when a collective environmental target is not met and subsidizes them when the group performs better than the target, individual group members will face incentives to behave efficiently if the tax/subsidy rate is set equal to the marginal damages from ambient pollution (Segerson 1988; Kotchen and Segerson 2018).

Second, free-riding can be eliminated if firms can be excluded from receiving the group-based benefit when they do not contribute to meeting the group standard, although using exclusion to reduce free-riding requires the ability to monitor the practices or performance of individual group members. Eliminating free-riding through this form of exclusion underlies the concept of a "green club," in which only contributors reap the benefits of club membership (Potoski and Prakash 2005; Kotchen and van't Veld 2011). Exclusion can also be used by cooperatives as a way to reduce or eliminate free-riding, as, for example, when fishing cooperatives revoke membership for vessel owners who violate rules for sharing costs or benefits (e.g., Holland 2018).

Finally, even if some firms free-ride in the sense of not contributing toward meeting the collective target, this does not necessarily mean that the target will not be met. As long as the benefits of meeting the target are sufficiently large for enough members of the group, these

[5]This is analogous to the classic free-riding that results in under-provision of a public good.

[6]This implies that the policy instrument can be designed to yield a first-best outcome (i.e., an outcome that maximizes social welfare). For a detailed discussion, see, for example, Segerson (1988) and Xepapadeas (2011).

firms may still undertake the actions necessary to ensure the target is met, despite the free-riding by others (Dawson and Segerson 2008). In this case, the program can still be successful in terms of achieving the collective performance goal, but, as noted above, the target will generally not be met at least cost.

Incentives to Coordinate (or Collude)

In some cases, ensuring that collective environmental goals are met efficiently requires coordination among members of the group. To illustrate, suppose that a group faces a fixed penalty (like closure of a fishery) if aggregate harvest or bycatch exceeds an allowable limit. If each vessel owner expects that all other vessels will contribute to staying within the collective limit, then it may be in each vessel's interest to contribute to that effort as well. However, if a vessel owner expects that other vessels will not contribute, then it may not be in the individual vessel owner's interest to contribute unilaterally. To ensure that the first outcome (the "good" equilibrium) rather than the second outcome (the "bad equilibrium") occurs, the vessel owners need some way to coordinate so that each knows that others will contribute sufficiently toward the collective goal.

Note that in the example above, there are multiple possible equilibrium outcomes under which the target is met, but they are not all equally "good." In fact, only one of them is an outcome where each member of the group contributes the efficient amount, which means that the target is met at the lowest possible cost. Under other outcomes, the target is met but not at least cost, due to the second type of free-riding discussed above (where some firms "oversupply" and others "undersupply"). Coordination within the group can help to ensure that the most efficient outcome (where the target is met at least cost) emerges.

There is a large literature on self-governance of common-pool resources that identifies factors that determine whether attempts at coordination are likely to be successful (Ostrom 1990, 1994). Those same factors are also likely to be important determinants of whether coordination leads to a good or a bad equilibrium when group policies with more than one equilibrium are implemented. One important factor is how the group is organized, and how it controls or influences decisions by group members. Groups may be composed of autonomous members, with each making independent decisions, or they may have a central decision-making body that sets general "rules" under which the group will operate, including what (if any) restrictions members will face in making their own choices. The rules could even specify the allowable actions of individual members of the group, such as how much land individual landowners within the group can clear or when and where individual vessels in a cooperative can fish. The design of these "internal" rules will play an important role in determining how the members of the group respond to the group policy. In addition, if payments (in the form of cash or other benefits) are made to the group rather than to individuals, the group must devise rules for how those payments will be distributed or used, taking account of both efficiency and distributional concerns. Perceived fairness of the rules for sharing costs and benefits is a key condition for successful coordination.

While coordination can help a group achieve a collective target, it can also create unintended incentives for collusion aimed at reducing aggregate tax burdens or increasing subsidy payments. This can lead to members of the group choosing outcomes that correspond to reducing pollution "too much" from an efficiency perspective (Hansen 1998).[7] For example, if the group receives subsidy payments for reducing pollution below some threshold and those payments continue to increase with further reductions, the group can benefit from those further reductions (if the additional subsidy payment exceeds any additional cost). Although these further reductions would lead to additional environmental improvements, such reductions would be inefficient if the cost of achieving them exceeds the corresponding social benefit. In such cases, coordination in the form of collusion would reduce rather than improve overall efficiency. All else equal, policies should be designed to encourage coordination in meeting a collective target without creating an incentive to collude and over-abate.

The discussion in this section has highlighted some general features of group-based policies and identified some of the issues and concerns that such policies raise. We turn next to the four specific contexts in which group policies have been—or could be—used by regulators seeking to improve environmental or resource management: nonpoint pollution (mainly from agriculture), PES programs, industry-wide voluntary approaches, and fisheries.

Nonpoint Pollution

Nonpoint pollution, most notably water pollution from agricultural land use, is one context where group-level policies have the potential to play an important role in achieving environmental quality goals. Farmers' activities, such as fertilizer use and manure management, contribute to high levels of nitrogen and phosphorous in streams, rivers, lakes, and coastal estuaries. However, as mentioned earlier, monitoring all farms to determine the contribution of each farmer to a given water pollution problem is difficult and costly, if not impossible. It is far easier and less costly to monitor the water quality in the affected waterbody. In fact, much of the early theoretical work on the use of group-based approaches to environmental and natural resource management was motivated by the difficulty of controlling nonpoint pollution using conventional (individual level) policy instruments (e.g., Segerson 1988).

Overview

The theoretical literature on the control of nonpoint pollution grew out of the industrial economics literature on "team" production. Team production refers to a situation in which the labor of workers leads to a jointly produced level of output and a manager can observe the joint output but not individual contributions or effort levels (e.g., Holmstrom 1982). With team production, rewards or penalties set by a manager must be based on the workers' joint output (e.g., whether the team meets its production targets) rather than the effort or output of individuals, because individual effort cannot be monitored. The nonpoint pollution context is analogous to team production. With nonpoint pollution, the contributions of individual polluters combine to determine a level of environmental quality, and a regulator can observe overall environmental quality but not the individual contributions to it. The regulator can then set rewards or penalties tied to the "joint" production of water quality.

[7] This is analogous to the detrimental effects of collusive behavior in antitrust contexts, where firms collude to restrict output in an effort to increase profits.

The theoretical literature on nonpoint pollution has examined a number of specific policies based on group performance, where group performance is defined relative to a specified target level of ambient water quality. Under these policies, the target can be coupled with (i) rewards paid to each farmer when the group meets the target (a pure subsidy approach); (ii) penalties imposed on each farmer if the group fails to meet the target (a pure tax approach); or (iii) a combination of both (a tax and subsidy approach). The amount of the tax or subsidy can be fixed or it can vary proportionally with the amount by which the target is exceeded or missed. As noted earlier, by setting the tax or subsidy rates equal to the marginal social costs of pollution and setting the target at the socially efficient level, regulators can, in theory, overcome free-riding and induce individual farmers acting independently to invest efficiently in pollution abating activities (Segerson 1988; Xepapadeas 2011; Kotchen and Segerson 2018).

An Example

Despite the extensive theoretical literature on the efficient design of group-level policies in this context, these policies have rarely been used in practice to control agricultural nonpoint pollution. We are aware of one example—the Florida agricultural privilege tax—which is levied on agricultural land in the Everglades Agricultural Area. The tax rate is reduced (through a tax credit) when aggregate phosphorous loadings from the basin are reduced below a target threshold, thereby rewarding farmers for basin-wide water quality improvements that go beyond the threshold (Hoffmann, Boyd, and McCormich 2006; Daroub et al. 2011). However, this collective reward is not the sole mechanism in place. It is coupled with mandatory requirements and individual incentives for the implementation of best management practices designed to meet the target threshold. Because these requirements and incentives were implemented at the same time as the agricultural privilege tax, it is difficult to isolate the impact of the tax itself.

Evidence

Because real-world use of taxes and subsidies based on ambient pollution is so limited, what we know about their effectiveness comes primarily from laboratory and field experiments. Most of the laboratory studies ask student subjects to make choices that mimic pollution abatement decisions when faced with different policies that are based on group outcomes. Laboratory experiments are, of course, only imperfect predictors of what can be expected in the real world. However, Spraggon and Oxoby (2010) argue that real-world outcomes might actually be more efficient than those that emerge in laboratory settings, because outcomes improve when decision-makers have information about and understand profit-maximizing choices, which would be more likely in the real world than in the laboratory.

Laboratory Experiments

The results from laboratory experiments that examine the impacts of these group policies are generally consistent with what theory predicts (Giordana and Willinger 2013). For example, policies based on proportional tax incentives (or a tax/subsidy combination) tend to be efficient, although those that involve subsidies may lead to collusion and over-abatement. In addition, even when policies result in a given environmental target being met, that target is not necessarily met at least cost (i.e., some subjects over-abate and others under-abate). Fixed (as opposed to proportional) group penalties tend not to perform well, presumably because of the difficulty in coordinating to reach the preferred equilibrium in a setting in which individuals are each acting on their own (i.e., when there is no communication). Communication can lead to more coordination and hence more efficient outcomes, but also to greater collusion, which can reduce efficiency (Vossler et al. 2006; Suter, Vosseler, and Poe 2008). Thus, in settings where members of the group are encouraged to communicate to improve coordination, regulators should anticipate possible collusion when setting penalties and/or rewards.

Field Experiments

In addition to the laboratory experiments, there have been a small number of field experiments that have examined the impacts of a collective performance approach to reducing agricultural pollution. In one field experiment, farmers in a West Virginia watershed were given the opportunity to participate in a scheme where the group of participants received a collective performance-based payment that depended on collective nitrate-nitrogen loadings (Collins and Maille 2011).[8] In contrast to the mechanisms tested in the laboratory experiments, where payments went directly to individuals, in this field experiment, the payment went to the group as a whole. The group then had to decide how to distribute the payment among its members. Interestingly, Collins and Maille (2011) find that the group chose to manage the distribution of payments in a very collaborative/cooperative manner that considered both incentives to participate and the cost and likely effectiveness of pollution abatement by individual farmers. In addition, the collective approach created incentives for the collection of information about pollution sources within the watershed, which could then be used to better manage pollution reduction efforts.

Taxes based on group performance have also been field-tested in other agricultural contexts where monitoring individual behavior or contributions is difficult. For example, Reichhuber, Camacho, and Requate (2009) studied the effectiveness of collective tax and tax/subsidy approaches in the context of biodiversity and the over-harvesting of nontimber products by Ethiopian farmers. Consistent with theoretical predictions from the nonpoint pollution literature, they find that both policies achieved a high level of aggregate efficiency.

Collective PES Programs

PES programs have been promoted in a wide variety of contexts as a means of creating conservation incentives, particularly in developing countries. Examples include paying farmers to reduce deforestation or improve water quality. Payments can be based on practices that are undertaken (practice-based PES) or on observed environmental outcomes (performance-based PES). Most applications, such as the U.S. Department of Agriculture's Conservation Reserve Program (Wu 2000), involve contracts with and incentive payments to individual landowners. However, some PES programs rely on group-level contracts, because land is

[8]Because participation in the experiment was voluntary, the program was similar to the PES programs we discuss in more detail in the next section.

jointly owned or managed or as a means to reduce contracting, monitoring, and enforcement costs (Kerr, Vardhan, and Jindal 2014).

Overview

As with all group-level policies, the positive and negative incentive effects of collective PES programs depend both on how the program is designed and how the group is organized and operates. With respect to program design, all collective PES programs include some form of collective responsibility, which typically means that payment will not be received unless the level of group practices or performance reaches a given target. However, the form of the "payment" or reward for meeting the target can vary (Engel 2015). For example, it can be a cash payment or an in-kind payment (i.e., a transfer of resources such as seed or fertilizer). In addition, the payment can be made directly to individuals (based either on the group's overall contribution or on the individual's contribution, if observable) or it can be made to the group (e.g., the community or village). In the latter case, the group must decide how the payment is used or distributed. The group could, for example, decide to distribute the payment to group members in some agreed-upon way or to use the payment for some purpose that would benefit all group members, such as investment in local public schools, infrastructure, or community-owned capital equipment (e.g., Munoz-Pina et al. 2008).

Examples

There are numerous real-world examples of collective PES programs, many of which have involved a resource that is commonly owned or used by a village or community. For example, in Ecuador, communities with communal land rights can receive payments for limiting grazing and other ecologically damaging activities on their lands. Although payments are also available to individual landholders, most of the land enrolled in the program is communal (Hayes, Murtinho, and Wolff 2015). Similarly, a large share of Mexico's payments for hydrological and biodiversity services goes to communities that collectively own and manage enrolled land (Munoz-Pina et al. 2008; Garcia-Amado, Perez, and Garcia 2013; Costedoat et al. 2015). In Sweden, reindeer herder villages with communal grazing rights receive payments for the conservation of wolverines and lynx based on the number of offspring observed in an annual inventory (Zabel, Bostedt, and Engel 2014). These performance-based payments directly reward conservation that is based on population outcomes, which are observable only at the village level. Under Japan's Farmland, Water and Environmental Conservation Improvement Scheme, self-formed "activity groups" can receive payments for collective conservation efforts that reduce negative environmental impacts from agriculture (Ito et al. 2018). In this case, contracting with a group rather than individuals both reduces transaction costs and incentivizes cooperation.

Evidence

In practice, it is often difficult to identify the extent to which a PES program generates *additional* conservation benefits beyond what would have occurred otherwise (Pattanayak, Wunder, and Ferraro 2010; Börner et al. 2017). Nonetheless, several observational studies have examined the impact of collective PES programs and there is also evidence from field experiments.

Observational Studies

Observational studies of collective PES programs evaluate how these programs affected either the actions undertaken by individuals or the group or the resulting environmental outcomes. For example, in their study of collective payments to Ethiopian farmers, Hayes, Murtinho, and Wolff (2015) find evidence suggesting that communities that participated in the program strengthened their grazing restrictions after participation. Although the authors caution against drawing conclusions about causation, their evidence is consistent with theory (discussed earlier) that predicts that collective payments can create incentives for groups to develop internal rules for cooperation. Similarly, Ito et al. (2018) find that communities that had previous experience with collective action were more likely to participate in a collective PES program and that, after controlling for prior differences between participants and nonparticipants, communities that participated in the program were more likely to carry out collective conservation activities. Finally, in a study of the impact of payments for biodiversity conservation in Mexico, Costedoat et al. (2015) find that communities that participated in the PES program had lower deforestation rates than nonparticipating communities.

Field Experiments

Researchers have also used field experiments to examine how collective payment or reward systems affect individual decisions. For example, Narloch, Pascual, and Drucker (2012), Midler et al. (2015), and Narloch, Drucker, and Pascual (2017) conducted a number of experiments with farmers from the Bolivian and Peruvian Andes. In these studies, groups of farmers signed contracts that would reward them with cash or in-kind payments if, and only if, all farmers in the group planted traditional (environmentally friendly) crops on the agreed-upon acreage. The experiments compared different payment designs, including individual and collective rewards. A general conclusion that emerges from these studies is that group-level contracts can create strong incentives for conservation when payments for meeting group targets are made directly to individuals, or when payments are made to the group and the group is able to develop trust among members and communicate and deliberate about collective action.

Similarly, Kaczan et al. (2017) used a field experiment in Mexico to examine the impact of a collective PES program on forest conservation. Consistent with the three studies just discussed, they find that a collective PES program can have an impact on conservation effort by individuals within the group, and that the level of individual contribution depends on the existence of within-group coordination mechanisms. This conclusion is also consistent with the field experiment results in Rodriguez, Pfaff, and Velez (2018), who studied collective reward systems for small-scale gold miners in Colombia and find that the existence of trust within the group was a key factor in determining the effectiveness of collective payment schemes.

Industry-Wide Voluntary Approaches

Voluntary approaches to improving environmental quality are often viewed as viable alternatives to command-and-control or more centralized market-based approaches (Kotchen 2013; Segerson 2013). Under a voluntary approach, a participating individual or firm voluntarily agrees to take steps to reduce pollution or increase conservation in response to some inducement. The PES programs we have just discussed are one type of voluntary approach. In the case of the PES programs, the inducement to participate comes from the payments that participants receive, which is a "carrot" approach to inducing participation. However, in many cases, the inducement comes from a "stick" approach; that is, the threat of regulation. Firms can also undertake voluntary environmental improvements as a way to achieve some perceived market-related gain, such as improved corporate reputation, product differentiation, or market access.

Overview

In most cases, government-sponsored voluntary programs target individual firms or landowners.[9] However, in some cases, rewards or penalties target an entire industry. This occurs, for example, when the government threatens to regulate an entire industry if the industry does not self-regulate—that is, if firms within the industry do not achieve pollution reduction goals on their own. In this case, the level of industry performance that is necessary to avoid regulation is effectively a group performance standard, with a penalty to all members of the group (i.e., all firms in the industry) if the standard is not met. Alternatively, an entire industry might benefit from self-regulation, through, for example, industry-wide reputational gains. In either case, the industry's performance overall affects the costs, benefits, or both of individual firms within the industry.

When regulatory threats or reputation gains occur at the industry level, the potential for free-riding arises. The incentives to free-ride can be reduced or eliminated if firm-level decisions or outcomes are observable so that "good" actors can be distinguished from "bad" actors. For example, if a regulator threatens to impose a tax or regulation on an entire industry for failure to self-regulate emissions, the regulator could provide some tax or regulatory relief to individual firms that can demonstrate sufficient reductions in their own emissions (e.g., Dijkstra and Rubbelke 2013). Similarly, when self-regulation leads to reputational gains, free-riding can be reduced if firms within an industry realize those gains only when they have demonstrated through an external "certification" process that they are environmental stewards (Kotchen and van't Veld 2011). In both of these cases, free-riding is addressed by excluding nonparticipants from reaping the benefits of voluntary improvements.

Examples

There are numerous examples of industry-wide regulatory threats. In the 1990s, the U.S. metal finishing industry faced the threat of new regulations on wastewater discharges; this ultimately led to the development of the voluntary Strategic Goals Program, an effort

[9]Numerous studies have examined the effectiveness of this type of voluntary program and found mixed results (e.g., Morgenstern and Pizer 2007; Borck and Conglianese 2009).

designed to forestall new regulation by promoting voluntary actions aimed at reducing pollution (Brouhle, Griffiths, and Wolverton 2009). Similarly, many U.S. electric utilities faced with the possibility of industry-wide regulation of greenhouse gas emissions sought to forestall regulation by participating in the U.S. Department of Energy's voluntary Climate Challenge program, which encouraged voluntary emissions reductions (Delmas and Montes-Sancho 2010). In these and other examples, a group of firms in a given industry faced a collective threat that would be triggered (with some probability) if the industry did not improve its performance on its own.

As discussed earlier, industry self-regulation can also be motivated by a desire to improve the industry's reputation. For example, the desire to improve the chemical industry's public reputation led to the creation in 1989 of Responsible Care, a voluntary program designed to improve the environmental and safety performance of firms in the Chemical Manufacturers Association and thus the industry's public image (King and Lenox 2000). Industry self-regulation can also be profitable for a group of firms when it affects market power or structure. For example, the decision by top European appliance manufacturers to voluntarily eliminate production of low energy efficiency (i.e., high polluting) washing machines likely increased their profits, at least initially, when commitment and compliance with the agreement were high (Ahmed and Segerson 2011).

Evidence

Empirical evidence on the impact of industry-wide regulatory threats and reputational gains on voluntary pollution control is consistent with theoretical models of industry-wide penalties/rewards. Those models predict that some firms will still participate even if others do not—that is, even if there is free-riding by some firms (e.g., Dawson and Segerson 2008). For example, in his study of the Responsible Care Program, Lenox (2006) finds that although there were incentives to free-ride, most firms were better off with than without the program, and thus some firms were willing to participate, even though others did not. Similarly, Delmas and Montes-Sancho (2010) find that participation in the Climate Challenge program was high; in fact, by the end of the program, utilities that had been responsible for more than half of the industry's carbon emissions in 1990 had participated.

However, a high level of participation does not necessarily mean that a program has been effective. Most empirical studies of voluntary approaches have estimated their effectiveness by comparing the environmental performance of participants and nonparticipants. King and Lenox (2000) find no evidence that firms within the industry that joined the Responsible Care program had better environmental performance than those that did not. Similarly, Delmas and Montes-Sancho (2010) find that utilities that participated in the Climate Challenge program did not reduce emissions more than nonparticipating utilities. These conclusions must be interpreted with some caution, however, because estimates of program impacts that are based on comparisons of participants and nonparticipants will be biased if some of the program's impacts spill over to nonparticipants (Zhou, Bi, and Segerson 2020).

Moreover, even if the impact of a voluntary program can be accurately estimated, it is difficult to identify the specific role of a potential industry-wide threat or reward. For this reason, the empirical evidence regarding the effectiveness of industry-wide threats or rewards is both rather limited and mixed (Segerson 2017). For example, in their study of the metal finishing industry and the Strategic Goals program, Brouhle, Griffiths, and Wolverton (2009) find that both participants and nonparticipants reduced emissions in response to the threat of regulation, suggesting that the threat of regulation was effective even if the voluntary program itself was not. In contrast, Harrison and Antweiler (2003) find that the threat of regulating releases of toxic substances under the Canadian Environmental Protection Act had relatively little impact on releases, which were driven primarily by existing regulations rather than the threat of future regulation.

Fisheries

Historically, fisheries management has been based on regulations such as limitations on the number of vessels that can fish, the number of days that fishing is allowed, or the type of gear that can be used. The inefficiencies that these regulations create are well-recognized, which has led to an increased interest in the use of rights-based fisheries management, under which vessels are granted quotas that give them rights to a certain amount of harvest, incidental bycatch, or even habitat impact.

Overview

Under rights-based management, rather than allocating rights to individual vessels, regulators can allocate rights or quotas collectively to groups of vessels (Abbott and Wilen 2009; Zhou and Segerson 2016; Holland 2018). For example, harvest quotas (or a certain share of the total allowable catch) can be granted collectively to a fishing cooperative comprised of multiple vessels, which must then manage its quota and establish rules for how members will collectively keep their total harvest within the cooperative's allowable limits. Similarly, limits for bycatch can be imposed at a group level rather than on individual vessels. For example, a bycatch limit could be set for an entire fleet or fishery rather than for individual vessels.

Whether applied to harvest, bycatch, or habitat, collective or group limits are analogous to the group performance mechanisms discussed above, whereby a penalty is triggered when the group exceeds its allowable limit. In this case, the penalty is typically closure of the fishery. While less common, an alternative is to allow the group to purchase additional quota when it exceeds its limit (Stewart and Leaver 2016). The threat of a fishery closure or the cost of having to buy additional quota provides an incentive for the group to devise internal rules to manage its quota effectively. In addition, given that both harvests and bycatch are inherently stochastic (i.e., they cannot be completely controlled by fishers), issuing a quota to a group of vessels rather than to individual vessels can also provide a mechanism for pooling the risks that any individual vessel might catch more than expected and thus exceed a vessel-level limit (Holland and Jannot 2012; Zhou and Segerson 2016; Holland 2018).

Examples

There are many examples of fishing cooperatives, with many of them facing collective limits on harvest and/or bycatch (Deacon 2012; Holland 2018). Ovando et al. (2013) find that approximately 50 percent of the fishing cooperatives they surveyed faced a government-imposed total allowable catch, and, according to Bonzon et al. (2013), 8 percent of the single-species and 23 percent of the multi-species catch-share programs worldwide allocate quota to a group. For example, in 2002, a share of the total allowable catch in the Alaskan Chignik salmon fishery was allocated to a voluntarily formed cooperative that was allowed to independently manage its own share (Deacon, Parker, and Costello 2013). Similarly, in 2010, the New England groundfish fishery introduced a system under which vessel owners were allowed to form groups (called "sectors") that would receive a collective allocation of allowable catch (Holland and Wiersma 2010). The Bering Sea and Aleutian Islands groundfish fishery has also relied on a group-based approach, in which quotas for target species were initially set at the fleet level but were subsequently allocated to individuals who could pool their quota to form cooperatives. These same cooperatives were then allocated shares of the bycatch limit for nontarget protected species (Abbott, Haynie, and Reimer 2015).

Although group-level limits are more commonly used to limit the bycatch of other fish species, they can also be used for the bycatch of nonfish species. A notable example involving endangered species is the Hawaiian longline swordfish fishery, which since 2004 has been subject to fleet-wide limits on the total allowable number of loggerhead and leatherback sea turtle interactions, with closure of the fishery if/when either aggregate limit is reached (e.g., WPFMC 2018).

Evidence

There is evidence of fishing cooperatives successfully managing collective rights based on case studies in the United States and throughout the world (Townsend, Shotton, and Uchida 2008; Holland 2018). Coordination has been shown to be effective in reducing the "race to fish," providing public goods for the group (such as information about bycatch hotspots or technology development), enforcing gear restrictions, enhancing product quality, protecting habitat, and reducing financial risks through pooling revenue or quota (Deacon 2012; Holland 2018). However, success appears to depend on the ability of the group to resolve internal and external conflicts and monitor and enforce within-group rules to limit free-riding (e.g., through internal fines, social pressure, or threats of exclusion). Case studies suggest that the ability to limit free-riding is enhanced when external observers are on vessels to monitor harvest and/or bycatch, members of the group have binding contracts committing them to the rules, and social pressure is effective (Holland 2018). Simply setting a fleet-wide total allowable catch, without having mechanisms to establish and enforce within-group rules, is unlikely to be effective (e.g., Abbott, Haynie, and Reimer 2015).

Although there is extensive case study evidence concerning the impacts of collective rights, there have been few quantitative assessments of those impacts. Quantifying the impacts of collective quotas faces many of the same empirical challenges that arise for PES programs, including the difficulty in establishing a valid baseline for comparison and the fact that fishers often voluntarily join cooperatives, which leads to potential self-selection problems. Nonetheless, the limited quantitative evidence suggests that collective quotas can be effective in inducing behavioral changes. Because these quotas are enforced through regulation (typically, closures), the question of interest in this case is not whether the group has met the collective limit, but rather whether it has met that limit in a more efficient way than under an alternative regulatory approach. In the case of the Alaskan

Chignik cooperative, Deacon, Parker, and Costello (2013) find that the vessels that joined the cooperative selected the most efficient among them to harvest the group's allowable catch, thereby maximizing rents for the entire group. In fact, the authors conclude that the creation of the cooperative led to a gain in economic rents of at least 33 percent. Abbott, Haynie, and Reimer (2015) examine the impact of moving from fleet-wide collective limits for both target and bycatch species in a Bering Sea and Aleutian Islands fishery to a system where voluntarily formed cooperatives could collectively manage their allowable catch. They find that the formation of groups to manage quota caused behavioral changes that led to significant improvements in the fishery, including adjustments in the timing and location of fishing activities to allow target species to be harvested with less bycatch. Huang et al. (2018) find evidence of other behavioral responses (such as increases in effort per vessel and changes in fishing locations) by some types of vessels that joined cooperatives under the New England groundfish sector program.

National Ambient Regulations

The preceding examples highlight group-level policies imposed on a specific geographic area, sector, or both. Two of the most significant national environmental policies in the United States—the CAA and the CWA—illustrate the use of group-level polices on a much broader scale, through federally imposed ambient air or water quality standards.

Ambient Standards under the CAA and CWA

A central component of the CAA is the setting of National Ambient Air Quality Standards (NAAQS), which apply to six common air pollutants.[10] States are responsible for ensuring compliance with these standards by designated air quality control regions within their borders. For each pollutant, regions are categorized as either in "attainment" or in "nonattainment" depending on whether they meet the corresponding NAAQS. If an area is designated as nonattainment, states must submit to the EPA a State Implementation Plan (SIP) that presents a credible plan for bringing the area into attainment. If a state does not develop an SIP, or if the SIP is not approved, the CAA gives regulators the authority to impose sanctions that would be costly to both the region and individual polluters, such as losses in access to federal grant funding and/or restrictions on permitting new facilities.

The CWA requires that states and territories maintain a list of waterbodies within their jurisdictions that do not meet applicable water quality standards. In addition, a state must develop a Total Maximum Daily Load (TMDL) analysis for each pollutant, which represents a plan for meeting national ambient water quality standards for a given waterbody. TMDLs define the maximum total amount of pollution (i.e., the pollution "load") that the waterbody can receive and still meet those standards. This total allowable load is then allocated across all sources that contribute to polluting the waterbody, including both point and nonpoint sources.

[10]The six pollutants are ground-level ozone, particulate matter, carbon monoxide, lead, sulfur dioxide, and nitrogen dioxide.

NAAQS versus TMDLs

Both NAAQS and TMDLs involve ambient standards, which define a collective performance standard for the polluters within a given region. Nonetheless, there are some important differences between NAAQS and TMDLs. First, many of the sources of the air pollutants governed by NAAQS are point sources, while the sources contributing to the water pollution problems that TMDLs are designed to address are often nonpoint sources. In the case of NAAQS, there are enforceable penalties that can be imposed on regions for failure to meet the ambient standards, and states have regulatory authority for controlling the behavior of facilities within the region (e.g., for limiting power plant emissions). In this sense, the state can effectively act like a centralized collective that coordinates compliance within the air quality control region, setting the rules to control within-group behavior and facilitate compliance with ambient standards. In contrast, TMDLs do not include clear penalties for failure to meet the group limits or enforceable rules to control the behavior of all contributing polluters. This is due, at least in part, to the fact that the CWA does not have a general regulatory structure for limiting nonpoint pollution, which is a primary source of pollution for many waterbodies. Uncontrolled nonpoint pollution is much less of a concern for air pollution, which stems primarily from point sources or vehicle emissions, both of which are regulated under the CAA.

Given these differences and the findings from the theoretical and empirical literature on collective approaches discussed earlier, it is not surprising that NAAQS appear to be more effective than TMDLs. The CAA, along with the NAAQs provision, is generally viewed as having been an effective environmental policy, leading to significant reductions in air pollution over the past several decades (EPA 2017). In contrast, there is no strong evidence that TMDLs have been broadly effective in improving water quality thus far. The evidence about TMDLs is limited, at least in part because of a shortage of monitoring data and the fact that many TMDLs were developed recently, which means it may take time for their impacts to be known (Norton et al. 2007). However, it is possible that even with more data and time, TMDLs will still prove insufficient to meet water quality goals. One reason for this is that in many cases, their implementation relies on voluntary efforts by farmers, an approach that is unlikely to be effective unless coupled with strong incentives or requirements to curb polluting activities (Segerson 2013).

Summary and Conclusions

To date, environmental and natural resource economics has generally focused on policies aimed at individuals, such as individual firms, landowners, or vessel owners. However, there are a number of reasons why, in some contexts, policies might be applied at the group rather than the individual level, including lack of observability of individual actions, the need for coordination, group ownership of resources, or simply lower transaction costs. This article has provided an overview and synthesis of the use of group policies in a number of different contexts, with the aim of better understanding how these policies affect behavior and, ultimately, effective environmental and resource management.

Our main findings and conclusions can be summarized as follows. The theoretical literature shows that, by appropriately setting rewards or punishments, regulators can design group-level policies that overcome free-rider incentives and effectively meet environmental and natural resource management objectives. This over-arching finding is also supported by the empirical evidence, which comes from a variety of contexts and is based on a range of empirical methods, including laboratory experiments, field experiments, and both case study and statistical evidence from real-world experience.

Indeed, the literature suggests that to be effective, group-level policies must include rewards and penalties that provide strong incentives for groups to meet targets in a cost-effective way. Moreover, to the extent that meeting targets cost-effectively requires coordination within the group, it is also important for the group to have or create within-group institutions or mechanisms to facilitate and enforce that coordination. More specifically, our review suggests that the following three conditions must be met for group-level policies to be effective.

Strong External Incentives

Group-level policy must establish a strong set of *external* incentives for the group as a whole to meet the collective target. These incentives can be based on either "carrots" or "sticks," but they must be sufficiently strong—that is, there must be an adequate regulatory structure to impose costly penalties or the necessary funds to finance sufficient rewards. For example, if rewards or penalties are fixed, the penalties or rewards must be sufficiently strong so that if some members of the group free-ride, other members will still find it beneficial to take the necessary actions to ensure that the overall group target is met. Moreover, incentives will be effective only if there is adequate enforcement of penalties or follow-through on rewards, which will often require external monitoring of group performance by a regulator or third party. Indeed, the lack of an adequate enforcement mechanism, such as an underlying regulatory structure, has hampered efforts to control nonpoint pollution, both at the local level and, more broadly, under the CWA's TMDLs. In contrast, the CAA, which has a strong regulatory structure for enforcing ambient air quality standards, has been more successful.

Incentives Designed for Cost-Effectiveness

The incentives must be carefully designed to ensure that targets are not only met but also met cost-effectively. Regulators can design group rewards and penalties—and set tax rates and subsidies—in a variety of ways. For example, when penalties and rewards are proportional to how far the group exceeds or falls short of a collective target, the per-unit rate should be set to ensure that individual group members face the full marginal social costs and benefits of their actions, which, in principle, eliminate incentives to free-ride. Indeed, the empirical evidence (mostly from laboratory experiments) generally indicates that such policies are efficient.

Coordination

Targets are more likely to be met in a cost-effective way if mechanisms are in place to foster coordination among group members. For example, when penalties or rewards are fixed (as, e.g., with the closure of a fishery or imposition of a costly regulation), multiple equilibria are possible, including outcomes under which the target is not met or is not met cost-effectively, and the challenge is to facilitate coordination to ensure a “good” outcome rather than a “bad” outcome. Similarly, if the group uses a common property resource such as a fishery, coordination concerning where, how, and when individual members access the resource can affect the overall return to the group. The ability of the group to coordinate will typically depend on the group having or developing internal operating rules that promote efficiency, as well as the group having the internal (within-group) capacity to monitor and enforce those rules (Ostrom 1990, 2000). Enforcement could be based on either formal mechanisms (such as contracts within the group) or informal mechanisms (such as trust, social norms, and peer pressure). In addition, successful coordination depends on the group’s ability to design internal rules for allocating costs and benefits that are viewed as both fair and efficient.

Concluding Remarks

The findings and conclusions of our review suggest that, in certain contexts and when properly designed, group-level policies can be an important tool for regulators to use in managing environmental and natural resources. Moreover, we believe that the discussion and synthesis we have presented here provide an initial step toward a more unified view of the lessons that emerge from the theoretical and empirical literature about the use of group-level policies. Although the various contexts we have discussed are quite different, the principles that determine the likely success of group-level policies apply across them all. Thus, we believe that both researchers and policymakers interested in one context can gain insights by looking at the lessons learned about the effectiveness of these policies in other contexts. Our hope is that this article will encourage such cross-fertilization, thus helping to guide both future research and future policy.

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Abstract

Policies to address environmental and natural resource management are often implemented at the group level. The defining feature of such policies is that penalties or rewards are based on group rather than individual performance, or rights are allocated to a group rather than to individuals. This article discusses how group-level policies have been applied and studied across a variety of contexts in the literature on environmental and natural resource management. The aim is to identify common theoretical and empirical insights and lessons learned about the design and implementation of these instruments. A general finding is that group-level policies are most likely to be effective when rewards and/or penalties are designed to provide strong incentives for groups to meet targets in a cost-effective way. Moreover, to the extent that this requires coordination within the group, the effectiveness of policies will depend on whether the group has or can create its own institutions or mechanisms to facilitate and enforce that coordination. (*JEL*: Q28, Q48, Q58)