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

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Abstract

Group-level policies are common in environmental and natural resource settings. The defining feature is that penalties or rewards are based on measures of group rather than individual performance, or the allocation of rights is to a group rather than to individuals. The existing research on group level-policies applies to specific areas of environmental and resource management, and this leaves a gap in the literature on cross-cutting themes about the design of these instruments and lessons learned about the successes and failures of their implementation. This paper reviews the way that group-level policies have been applied and studied in a variety of different environmental and resource settings. The aim is to identify broad theoretical and empirical insights that will help inform policy design and implementation. An overarching conclusion is that the effectiveness and efficiency of group-level policies depend not only on how the group-level instrument affects individual incentives, but also on the internal operating rules and norms of the groups themselves.

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Introduction

Market failure provides the lens through which economists understand environmental and natural resource problems. The existence of a market failure provides the economic rationale for policy interventions, and one aim of environmental economics is to identify the advantages and disadvantages of different policy instruments. The potential policy instruments that are typically considered fall into the broad categories of command-and-control regulations, market-based incentives, and voluntary- and information-based approaches. In most cases, the particular policies under study or used in practice target individual polluters or resource users, including the examples of emission controls at specific facilities, taxes on a firm’s inputs or outputs, and information disclosure about a company’s production practices or emissions.

In this paper, we consider environmental and natural resource policies that apply at the group rather than individual level. The defining feature of a group-level policy is that regulatory rewards or penalties are based on a group outcome rather than the practices or performance of individual agents, or rights are allocated to a group rather than to individuals. For example, regulations to protect air or water quality can be triggered by ambient levels of pollution rather than individual emissions. Other examples include payments for ecosystem services that are based on the collective performance of a community rather than individual landowners, and threats to impose costly regulations or taxes on an entire industry if it fails to collectively meet pollution control objectives through some form of self-regulation or voluntary pollution control. Similarly, fishing rights or bycatch limits can be allocated to a group of fishing vessels rather than to individual vessels.

While group-level policies are commonly employed and studied in a wide variety of environmental and natural resource settings, the literature provides little in the way of general insights about when and how group-level polices are an effective and efficient regulatory approach. There are a number of studies that focus on particular contexts, including agricultural nonpoint source pollution, payments for ecosystem services, industry-wide voluntary programs, and fisheries. Such context-specific research makes important contributions, yet a more unifying perspective that looks across these different settings is needed to more fully understand the potential and limitations of group-level policies as an effective form of environmental and natural resource management.

This paper reviews the way that group-level policies have been applied and studied in a variety of different environmental and resource settings. The aim is to identify broad theoretical and empirical insights that will help inform policy design and implementation. Rather than provide a comprehensive review, we focus attention on selected areas of environmental and natural resource management. In
particular, we consider group-level policies that have been applied to nonpoint source pollution, payments for ecosystem services, industry-wide voluntary program, and fisheries. We also compare and contrast group-level policies that arise through implementation of the Clean Air Act and the Clean Water Act based on ambient measures of air and water pollution.

The next section begins with a narrative description of the conceptual framework that guides our review. The framework draws on earlier theoretical work designed to provide a unified model for understanding the positive and normative consequence of group-level policies (Kotchen and Segerson 2018). We then organize the review into sections corresponding with each of the different environmental and natural resource settings. We conclude with a discussion of common themes and general findings related to the ways in which group-level approaches can, do, and should fit into the proverbial “toolbox” of policy instruments for environmental and natural resource management.

A Conceptual Framework for Group-Level Policies

Some environmental and natural resource settings are such that a group-level policy may seem like a potentially promising policy approach. There are at least four such settings. First, in some cases, individual actions may be difficult or impossible to observe, while an aggregate group outcome is easily monitored. For example, a regulator might find it difficult or prohibitively costly to monitor the activities of individual farmers that affect water quality, yet relatively simple to monitor the ambient level of water quality at the bottom of a watershed. Second, transaction costs might be lower when focusing at the group level. When seeking to promote ecosystem services such as deforestation, for example, it may be less costly to contract with a single entity representing a group rather than with many individual landowners. Third, group-level policies may provide an appealing approach if the natural resource under consideration is held under joint ownership of a group. For example, in many developing countries land is collectively owned or managed by a community or village. Finally, group-level policies may have advantages when spatial interactions among group members are important for determining outcomes, as, for example, when spatial externalities due to congestion create interdependencies among group members.

Group Definition and Externalities

Use of a group-level policy first requires a definition of the group, which in our context is a collection of agents whose economic activities can degrade the environment or use of a natural resource. For
simplicity, we will sometimes refer to the agents as firms, although they could also be individual landowners or vessel owners. The firms comprise a group because they represent a natural or feasible management unit based on biophysical interconnectedness (e.g., watershed), political jurisdiction (e.g., state or municipality), or economic sector (e.g., industry). Alternatively, a set of agents might voluntarily choose to come together to form a group, such as a group of fishing vessels that decides to form a fishing cooperative. In some cases, the payoffs of group members might already be interdependent, due, for example, to imperfect competition or within-group externalities stemming from spatial interactions or use of a common-pool resource. In other cases, in the absence of the policy, firms within the group might be quite independent in terms of the returns to their economic activities. For example, if they operate as perfectly competitive firms without any pre-existing within-group externalities, then each firm’s payoffs will initially be independent of other firms’ choices. However, introduction of a group-level policy will then, by definition, create a policy-induced interdependence.

When only within-group externalities exist, the group itself has an incentive to self-organize to address these externalities, and there is a large literature on how effectively and under what conditions they are likely to do so. Much of this literature is in the context of managing common-pool resources (Ostrom 1990). However, our interest in this paper is in contexts where there are outside-the-group externalities, i.e., externalities that arise when the economic activity of firms inside the group create social costs that are born by members of society outside the group. Although our primary interest is in the use of group-level policies to address externalities outside the group, we recognize that in some cases, both within- and outside-the-group externalities operate simultaneously. For instance, while fishing might generate an outside-the-group externality because of bycatch, the vessels within the fishery might also experience within-group crowding and intertemporal stock externalities. The existence of externalities outside the group means that the unregulated equilibrium will, in general, be inefficient, even if the within-group externalities are addressed. This is the market failure that provides the potential economic rationale for a policy intervention.

**Group-Level Instruments**

A group-level policy instrument differs from individual-based instruments in that each group member is rewarded or penalized based on the activities of the group as a whole. That is, under the policy, each member’s own costs, benefits, or both depend not only on its own actions but also the actions of all other group members. More specifically, a group-level policy is based on some observable and measurable indicator of group performance that is a function of all group members’ activities. Examples
include direct measures of environmental quality or impacts, such as total emissions, ambient concentrations, aggregate risks of spills or contamination, or a measure of ecological degradation such as regional deforestation. The aggregate measure of the group’s performance or activity then provides the basis for a policy-determined system of rewards or penalties that in turn affect individual incentives.

Several of the particular group-level mechanisms that might be employed have analogs based on individual behavior. Proportional taxes (with or without an allowable limit), proportional subsidies that depend on an established baseline, a combination of taxes and subsidies, and fixed penalties or fines for exceeding some limit, can all be based on either individual-based measures (if observable) or group-level measures. For example, a tax or subsidy could be based on emissions of an individual firm, or it could be based on the aggregate level of emissions of the group or the resulting ambient level of air pollution. Similarly, subsidies or payments for environmental improvements could be paid to individuals or groups (e.g., a village) and bycatch limits (with associated penalties for violation) could be set at the level of individual vessels or at a group level.

Cutting across the range of specific group-level policy instruments, however, are two key questions. First, can group-level policies create the individual incentives to meet intended levels of environmental quality or use of a natural resource? Second, are group-level policies cost effective, that is, do they maximize net benefits to group members conditional on meeting the intended group performance? The theoretical literature has identified a number of potential challenges to efficiency that can arise under group-level policies (Kotchen and Segerson 2018).

**Free-riding**

Free-riding may occur if some firms can benefit from an improvement in group performance even if they do not contribute to the cost of that improvement, or, equivalently, if the costs of abatement that a firm undertakes are borne fully by that firm while the benefits are shared by others within the group. These circumstances create an incentive for firms to free-ride on the efforts of other firms. Importantly, in our context free-riding can take two different forms. Under one form, all firms under-abate and hence, if there is a group-performance standard, it is not met. Under the second form, some firms under-abate, while others over-abate. In this case, a group-performance standard can still be met, but it will not be met at least cost.

There are several ways in which the potential for free-riding can be addressed. One possibility is through the appropriate design of rewards and penalties (Kotchen and Segerson 2018). For example, setting policy parameters to ensure that each group member faces both the full marginal social cost and
the full marginal social benefit of her activities will induce efficient decisions by all group members (Segerson 1988). Second, free-riding can be eliminated if firms that do not contribute to meeting the group standard can be excluded from receiving the corresponding benefits. Solving free-riding through this form of exclusion underlies the concept of a “green club,” where only contributors reap the benefits of club membership (Potoski and Prakash 2005; Kotchen and van’t Veld 2011). However, it requires the ability to monitor the practices or performance of individual group members. 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 from meeting the target are sufficiently large for enough members of the group, those firms will be better off taking the actions necessary to ensure the target is met, despite the free-riding of others (Dawson and Segerson, 2008). In this case, the program can still be successful in the sense of achieving the collective performance goal, but the overall cost of meeting the target would generally not be minimized.

Incentives for Coordination and Collusion

Even though there are ways to address the free-riding problem that can arise with group policies, efficient outcomes are nevertheless not guaranteed, nor are the conditions that give rise to them as straightforward as those for policies that focus on individual outcomes. One potential concern is the existence of multiple equilibria, which can arise under some policy designs. For example, if a firm expects that all other firms will contribute to meeting a collective goal, it might be in the firm’s interest to contribute as well. However, if the firm expects that no other firms will contribute, it might not be in the firm’s interest to contribute unilaterally.

Group-level policies may therefore create a policy-induced collective action problem centered on the need for coordination to meet compliance goals. When such coordination must arise from the bottom-up, the circumstances that are conducive may be similar to those that have been well-studied in the context of self-governance of common-pool resources (Ostrom 1990), and the extent or way in which members of a group coordinate can vary significantly. Groups can be comprised either of autonomous members each making independent decisions, or they can have a central decision-making body that makes decisions on behalf of the group members. Those decisions could dictate behavior by individual members of the group, such as how much land individual landowners within the group can clear. Alternatively, it could determine more general “rules” under which the group will operate, including what (if any) restrictions group members will face in making their own choices, and how any
payments that the group receives will be distributed or used. The nature and extent of within-group coordination can be a critical factor in determining the effectiveness of a group approach.

While coordination can help a group reach a better equilibrium, it can also create unintended incentives for collusion aimed at reducing aggregate tax burdens or increasing subsidy payments. In particular, members of the group might benefit from choosing outcomes that correspond to reducing pollution “too much” (i.e., beyond the level that balances marginal benefits and marginal costs). This will reduce overall efficiency unless the behavioral responses are anticipated by the regulator in the design of the policy mechanism.

Summary
The theoretical literature shows how the success of group-level policies will, in general, depend on not only how the particular policy instrument is designed (i.e., how the rewards and penalties are linked to the group outcome), but also on the internal operating rules and norms of the group itself. The experience with group-level policies discussed below confirms that these policies are most likely to be effective when the incentives are well-designed and there is an institutional structure or mechanism for coordination within the group.

Nonpoint Source Pollution
Water pollution from agricultural land use is one setting where group-level policies have the potential to play an important role. The activities of farmers, 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 previously, monitoring the activities on 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 ambient water quality in the affected waterbody. In fact, a focus on nonpoint source pollution spurred much of the early theoretical work on group-based approaches in an environmental context (e.g., Segerson 1988).

The theoretical literature on the control of nonpoint source pollution has its roots in the theoretical literature on team production, where 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). The nonpoint pollution context is analogous to this, although it is the pollution control efforts of farmers that produce a joint product, namely, water quality, and their individual
activities cannot be easily monitored. In the context of team production, rewards or penalties set by a manager are tied to the workers’ joint output, e.g., whether the team meets its production targets. For nonpoint pollution, it is a regulator who sets rewards or penalties tied to the production of water quality.

The literature has examined a number of specific ambient-based penalty/reward structures. All of these are based on a specified target level of ambient water quality, coupled with (i) rewards paid to each farmer when the ambient target is met (a pure subsidy approach), (ii) penalties imposed on each farmer when the target is not met (a pure tax approach), or (iii) a combination of both (a tax plus 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. By setting the tax or subsidy rates and the target appropriately, regulators can in theory overcome free-riding and induce individual farmers acting independently to invest efficiently in pollution abating activities (Segerson 1988; Xepapadeas 2011).

**Laboratory Studies**

Because of the lack of real-world experience with ambient-based policies, economists initially turned to the use of laboratory experiments to shed light on how these policies might be expected to work in practice. These studies test how well different policy designs work in a variety of different settings, such as with and without communication among group members. Most of the studies use students as subjects and ask those students to make choices that mimic pollution abatement decisions when faced with different policies based on group outcomes. Laboratory experiments are, of course, only imperfect predictors of what can be expected out “in the field.” However, work by Spraggon and Oxoby (2010) suggests that real-world outcomes might be more efficient than those predicted in the lab, since outcomes are improved when decision-makers have information about and understand profit-maximizing choices, which might be more likely in the field.

Nonetheless, the results from these lab experiments are generally consistent with theoretical predictions (Giordana and Willinger 2013). For example, policies based on proportional tax incentives (or a tax/subsidy combination) tend to be efficient, while those that involve subsidies can lead to collusion and over-abatement. In addition, in some experiments, even when the target is met, it is not met at least cost, i.e., some subjects over-abate while others under-abate. Fixed penalties for the group of subjects, which as noted above can generate multiple equilibria, tend not to perform well, presumably because of the difficulty of coordinating to reach the preferred outcome. Communication can lead to more efficient outcomes but also to greater collusion (Suter et al. 2008; Vossler et al. 2006).
Field-Based Studies

There have been a small number of field experiments involving a collective performance approach to reducing agricultural pollution. Field experiments use individuals who are actually a part of the context being studied. For example, for agricultural water pollution, the subjects used in the experiments would be farmers rather than students. In one field experiment (Collins and Maille 2011), farmers in a watershed in West Virginia were given the opportunity to participate in a collective payment scheme under which the group of participants would receive performance-based payments that were adjusted based on water quality levels (specifically, nitrate-nitrogen loadings) in the watershed. Because participation in the experiment was voluntary, the program was similar to the payment-for-ecosystem-services programs discussed in more detail below. However, in contrast to the mechanisms tested in the experimental literature, in this experiment the payment went to the group itself, which had to then decide how to distribute the payment to members of the group. Interestingly, the group chose to manage the distribution of group 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.

The ambient-based policies discussed here have also been field-tested in contexts that do not involve agricultural water pollution but share the key characteristics of nonpoint source pollution, namely, the difficulty of monitoring individual behavior or contributions. For example, Reichhuber et al. (2009) studied the effectiveness of collective tax and tax/subsidy approaches in the context of biodiversity and the over-harvesting of non-timber products (honey) by Ethiopian farmers. The results from these field experiments are consistent with results from the laboratory studies. In addition, they highlight the role of experience in building trust and expectations, which can in turn increase the likelihood of effective coordination to ensure that collective targets are met, a result that also emerges from the PES literature discussed below.

Real-World Experience

Real-world experience with ambient-based policies to control nonpoint sources of pollution is very limited. The one notable real world example, with actual penalties or rewards, that we are aware of is the Florida agricultural privilege tax, which is levied on agricultural land in South Florida and designed to improve water quality (through reduced phosphorous loadings) in the Florida Everglades (Daroub et al. 2011). The tax rate is reduced (through a tax credit) when aggregate phosphorous loadings are reduced,
thereby rewarding farmers for overall improvements in water quality. The legislation that established the tax, the Everglades Forever Act, also mandated the implementation of best management practices (BMPs) by growers within the affected districts. Thus, although phosphorous loadings fell significantly following implementation of the legislation (Daroub et al. 2011), it is not possible to determine what role the collective tax (as opposed to the BMP regulations) played in generating the reduction in phosphorous loadings.

**Collective Payment-for-Ecosystem-Services (PES) Programs**

PES programs have been promoted in a wide variety of contexts as a means of creating conservation incentives, particularly in developing countries. As discussed in the previous section, paying farmers based on improvements in water quality provides an example. Payments can be based on practices that are undertaken (practice-based PES) or on observed environmental outcomes, such as reduced deforestation (performance-based PES). Most applications involve contracts with (and incentive payments to) individual landowners. However, some PES programs involve group-level contracts (Kerr et al. 2014), which are sometimes used because land is jointly owned or managed, or as a means to reduce contracting, monitoring, and enforcement costs. A major concern is that group-level contracts and the associated payments will create incentives for free-riding, or will “crowd out” other pro-social or intrinsic motivations for undertaking conservation efforts.

As with all group-level policies, the positive and negative incentive effects of collective PES programs depend on both how the program is designed and the interactions/organization within the group. With regard to program design, although all collective PES programs include some form of collective responsibility, they can differ significantly in other dimensions (Engel 2015). Collective responsibility typically means that payment will not be received unless group practices or performance reach a given target. The “payment” or reward for meeting the target 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 this case, the group must decide how it is used. It 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). However, research using choice experiments to elicit
preferences over different payment mechanisms in Mexico and Tanzania has shown that individuals tend to prefer individualized payments over group rewards (Costedoat et al. 2016; Kaczan et al. 2013).

Examples of Collective PES Programs

Unlike the ambient-based policies for agricultural nonpoint source pollution, there are numerous real-world examples of the use of collective PES programs. For example, the Swedish performance payment scheme for conservation of wolverines and lynx makes payments to reindeer herder villages, where the payment amount is based on a measure of group performance, namely, the number of offspring observed in an annual inventory (Zabel et al. 2014). Similarly, Japan’s Farmland, Water and Environmental Conservation Improvement Scheme (CIS) makes payments to rural communities (rather than individual farmers) based on compliance with a contract that identifies collective efforts to maintain common-property resources such as irrigation facilities, canals, farmland and reservoirs (Ito et al. 2018). Other examples of community-based PES programs include: (1) village-level ecotourism programs in Cambodia, which make payments (in the form of revenue-sharing) to local villages for species conservation (Clements et al. 2010), (2) Mexico’s payments for hydrological and biodiversity services, a large share of which go to communities that collectively own and manage enrolled land (Munoz-Pina et al. 2008; Garcia-Amado et al. 2013; Costedoat et al. 2015), and (3) Ecuador’s payments to communities for limiting grazing and other ecologically-damaging activities on designated lands (Hayes et al. 2015).

Empirical Challenges

Evaluating the impact of any PES program, including collective PES programs, can be difficult because of the lack of an appropriate counterfactual to represent what would have happened in the absence of the program (Pattanayak et al. 2010). In addition, unlike collective taxes or penalties, which can be imposed on all members of an identified group, collective PES programs are often based on voluntary participation. This creates the potential for participation by individuals or groups that would have provided the environmental services anyway, which can “waste” available funds and limit program effectiveness. Identifying the extent to which the program generates additional conservation benefits beyond what would have been generated in the absence of the program requires the establishment of an appropriate benchmark. This is often difficult in practice. Participants can differ significantly from non-participants in ways that might affect what they would have done otherwise, which means that the performance of non-participants may not provide an appropriate benchmark when participation is
voluntary. This needs to be accounted for in the analysis of program impacts to reduce bias in the estimated impacts. Thus, rather than simply comparing “before” and “after” outcome measures, as is often done, reliable analysis of PES program impacts requires careful identification of and comparisons to the appropriate counterfactual benchmark using causal inference methods.

Evidence
A recent review by Borner et al. (2017) of PES program evaluations based on these methods finds some evidence of significant PES-driven environmental improvements, especially at more local levels. Some of the studies they review involve community-based PES programs. For example, using matching methods to estimate 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 non-participating communities.

Researchers have also turned to the use of experimental methods to test collective payment or reward systems in a lab or field environment. For example, a number of experiments involving collective payments have been conducted with farmers from the Bolivian and Peruvian Andes (Narloch et al. 2012; Midler et al. 2015; Narloch et al. 2017). 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 low-intensity traditional quinoa landrace varieties (rather than the more profitable but more environmentally-damaging commercial varieties) on the agreed-upon acreage. The experiments compared different payment designs, including individual rewards and collective rewards. These studies found that group-level contracts can create strong incentives for conservation when payments for meeting group targets are based on individual rewards, or when payments are made to the group and the group can communicate enough to develop trust and within-group collaborative deliberation.

Similarly, Kaczan et al. (2017) used a field experiment in Mexico to test the impact of a collective PES program for forest conservation. As with the previous studies, they find that a collective PES program can have an impact on conservation effort by individuals within the group, and that the level of contribution depends on the existence of within-group coordination mechanisms. This conclusion is also consistent with the experimental results in Rodriguez et al. (2018), who study collective reward systems for small-scale gold miners in Colombia and find that the existence of social capital is a key factor in determining the effectiveness of collective payment schemes.
Industry-Wide Voluntary Approaches

The PES programs discussed above fall within the broader class of policies known as “voluntary approaches.” Voluntary approaches to improving environmental quality are often viewed as a viable alternative to regulations and taxes (Kotchen 2013; Segerson 2013). All voluntary approaches require some inducement to participate or undertake voluntary initiatives. In the case of the PES programs discussed above, the inducement comes from the payments participants receive, which is effectively a “carrot” approach to inducing participation. However, for many voluntary programs the inducement comes from a “stick” approach, namely, the threat of regulation. Firms can also be induced to undertake voluntary environmental improvements for some perceived market-related gain, such as enhanced corporate reputation or product differentiation.

In most cases, government-sponsored voluntary programs target individual firms or landowners. An example is the U. S. EPA’s well-studied 33/50 Program (e.g., Bi and Khanna 2012). Numerous studies have examined the effectiveness of voluntary programs of this type (e.g., Morgenstern and Pizer 2007; Borck and Conglianese 2009). However, in some cases the benefit to an individual firm from voluntary action will depend not only on its own actions but also on the actions of other firms. This occurs, for example, when the government threatens to regulate an entire industry if the industry does not “self-regulate,” i.e., if the 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 prevent regulation constitutes 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 realize a reputational gain or reward from industry self-regulation. In either case, the industry’s performance overall affects the costs, benefits, or both of individual firms within the industry.

Examples

Numerous examples of industry-wide regulatory threats exist. In the 1990’s the U.S. metal finishing industry faced the threat of new effluent guidelines and pretreatment standards for wastewater discharges, which ultimately led to the development of the voluntary Strategic Goals Program, an effort designed to forestall the regulation (Brouhle et al. 2009). Similarly, many electric utilities faced with possible industry-wide regulation of greenhouse gas emissions sought to forestall regulation through participation in the U.S. Department of Energy’s voluntary Climate Challenge program (Delmas and Montes-Sancho 2010). In these and similar 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 collective performance on its own.

As noted, 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 reputation among the public 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 (CMA) and thereby improve 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 the principal 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 was high (Ahmed and Segerson 2011).

Free-riding with Voluntary Approaches

The group nature of regulatory threats or reputation gains creates the potential for free-riding. The general solutions to free-riding discussed above can be applied in this context as well. For example, if a regulator threatens imposition of a tax or regulation on an industry for failure to self-regulate, it could provide some tax or regulatory relief to individual firms that can demonstrate sufficient improvement in their own performance (e.g., Dijkstra and Rubbelke 2013). This can eliminate the incentive to free-ride, but requires that individual improvements be observable. Similarly, when self-regulation leads to reputational gains, free-riding can be reduced if firms within an industry only realize those gains when they are “certified” to have improved their environmental performance (Kotchen and van ’t Veld 2011). In both of these cases, free-riding is addressed by excluding non-participants from reaping the benefits of voluntary improvements.

When exclusion is not possible, free-riding can still be addressed through program design, but this becomes more complicated in this context because firms must anticipate how imposition of a threatened regulation (or loss of a subsidy or reputational gain) would impact them in the future when making decisions about voluntary abatement today. In other words, the policy should induce firms to make efficient decisions if/when the threat is imposed, but also needs to induce efficient decisions to avoid imposition of the threat in the first place. Appropriately designing the threat can create efficient incentives at both stages, but as in other contexts some policy designs will yield multiple equilibria (Segerson and Wu 2006). However, penalizing firms in the future based on not only whether they meet
a collective target but also on how close they come to meeting it solves the multiple equilibria problem (Suter et al. 2010).

Evidence
Empirical evidence on free-riding is consistent with the predictions of theoretical models of industry-wide penalties/rewards, which predict that some firms will still participate despite the lack of participation by others (e.g., Dawson and Segerson 2008). For example, in his study of the Responsible Care Program, Lenox (2006) finds evidence to support the hypothesis that, although there were incentives to free-ride, most firms were better off with the program than without it and hence some firms were willing to participate despite others not doing so. Similarly, Delmas and Montes-Sancho’s (2010) study of the Climate Challenge program finds that by the end of the program in 2000, utilities representing approximately 60 percent of the 1990 generation and utility-level carbon emissions had participated in the program.

However, with voluntary programs, a high level of participation does not necessarily mean that a program has been effective in changing behavior. Most empirical studies of voluntary approaches have estimated their effectiveness by comparing the environmental performance of participants and non-participants. For example, 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 join. Similarly, Delmas and Montes-Sancho (2010) find that utilities that participated in the Climate Challenge program did not reduce emissions more than non-participating utilities. These conclusions must, however, be interpreted with some caution, since estimates of program impacts based on comparisons of participants and non-participants will be biased if some of the program’s impacts spill over to non-participants (Zhou et al. 2019).

In addition, even if the impact of a voluntary program can be accurately estimated, it is difficult to identify explicitly the role that a potential industry-wide threat or reward played in changing behavior. For this reason, the empirical evidence regarding the effectiveness of collective threats or rewards is rather limited, and the limited evidence is mixed (Segerson 2017). For example, in their study of the metal finishing industry and the Strategic Goals program Brouhle et al. (2009) find that both participants and non-participants 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 regulation of toxic substances under the Canadian
Environmental Protection Act had relatively little impact on releases, which were driven primarily by existing regulations rather than threatened future regulation.

**Fisheries**

Group-level approaches have also been used in fisheries to manage harvest and bycatch levels. Rights-based management of fisheries arose in response to concerns about the inefficiencies of regulation. In many cases, rights in the form of quotas or catch shares have been given to individual vessels. However, rights can also be allocated to a group of vessels, such as a fishing cooperative, with the objective of addressing either within-group externalities (e.g., spatial or temporal congestion) or externalities outside the group (e.g., bycatch, habitat destruction, or impacts on recreational fishers) (Abbott and Wilen 2009; Zhou and Segerson 2016; Holland 2018). When allocated to a group, the group must manage its harvest quota and establish rules for how members will collectively keep the total harvest within the regulated limits.

Similarly, limits for bycatch can be imposed at a group level rather than on individual vessels. For example, an aggregate bycatch limit could be set for an entire fishery rather than for individual vessels. These limits could be applied to bycatch of non-target fish species (such as halibut in some fisheries) or bycatch of threatened or endangered species (such as sea turtles). Given that both harvests and bycatch are inherently stochastic, i.e., cannot be completely controlled by fishers, setting limits or issuing a quota for a group of vessels rather than individuals can also provide a mechanism for managing the risk of inadvertently exceeding individual quotas (Holland and Jannot 2012; Holland 2018).

Whether applied to harvest or bycatch, collective or group limits are analogous to the group-performance mechanisms discussed above. Group performance is determined by the total harvest or bycatch by the vessels within the group, and there is a penalty triggered when the group exceeds its allowable limit (typically, closure of the fishery). An alternative is to allow the group to effectively purchase additional quota that allows them to continue fishing by paying a fee or charge that is proportional to their exceedance, similar to what is done under New Zealand’s deemed value system (Steward and Leaver 2016).

**Examples**

There are many examples of fishing cooperatives (Deacon 2012), and many of them face collective limits on harvest and/or bycatch. A survey by Ovando et al. (2013) finds that approximately 50 percent of the
fishing cooperatives 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. As an 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 manage its share in its own way (Deacon et al. 2013). Similarly, in 2010 a new harvest management system was introduced into the New England groundfish fishery 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, where allocations were initially made at the fleet level and then to individuals who could form cooperatives. These cooperatives were then also allocated shares of a total allowable catch for non-target protected species (Abbott et al. 2015).

As noted, group-level bycatch limits can also be used for endangered species. A notable example 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., WPRFMC 2018).

**Evidence**

Examples of fishing cooperatives successfully managing collective rights can be found in the U.S. and throughout the world (Townsend et al. 2008; Holland 2018). There is also evidence of failures (see Ostrom 1990). Statistical analyses of the impacts of collective quotas face many of the same empirical challenges discussed above, including the fact that fishers often voluntarily join cooperatives, leading to potential self-selection problems. For example, in their study of the Alaskan Chignik salmon cooperative Deacon et al. (2013) find that the least skilled fishermen joined the cooperative, with more skilled fishermen opting to remain independent (i.e., in the common pool). Huang et al. (2018) also find evidence of self-selection. However, in contrast to the Chignik salmon cooperative, they find that trawl vessels that joined the New England groundfish sectors tended to be more efficient than those that stayed in the common pool. Thus, although both studies provide evidence of self-selection, they demonstrate that the incentives to join a group will likely be context-specific.

Regarding the impact of collective quotas, because these quotas are enforced through regulation (closures), the impact of interest in this context is not whether the group has met the collective cap or target, but rather how the cap has been met. In the case of the Alaskan Chignik cooperative, Deacon et al. (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. They estimated that the creation of the cooperative led to a gain in economic rents of at least 33 percent. In the context of a Bering Sea and Aleutian Islands trawl fishery, Abbott et al. (2015) examine the impact of moving from fleet-wide collective limits for both target and bycatch species to a system where voluntarily-formed cooperatives could collectively manage their allowable catch. They find that this change led to significant improvements in the fishery, including spatial and temporal adjustments to fishing effort for more efficient use of the quota for target species with less bycatch. Huang et al. (2018) also find evidence of other behavioral responses (such as increases in effort per vessel and changes in fishing locations) by trawl vessels that joined cooperatives under the New England groundfish sector program. However, joining a sector appeared to have had little impact on gillnet vessels or on the technical efficiency of either vessel type.

National Ambient Regulations

Two of the most significant environmental policies in the United States are the Clean Air Act (CAA) and the Clean Water Act (CWA). Importantly, both include ambient (group-level) environmental standards.

The CAA’s National Ambient Air Quality Standards

A central component of the CAA is the setting of National Ambient Air Quality Standards (NAAQS) for six criteria air pollutants. States are responsible for keeping designated air quality control regions within their borders in compliance. For each pollutant, regions are categorized as either in “attainment” or “nonattainment” depending on whether they meet the corresponding NAAQS. If an area is designated nonattainment, states must submit to the EPA a State Implementation Plan (SIP) for approval. The SIPs must demonstrate a credible plan for bringing the area into attainment. If a state does not develop a SIP, or if one is not approved, the CAA calls for sanctions that would be costly to both the state and individual polluters.

The CWA’s Total Maximum Daily Load Program

The federal CWA requires that states and territories maintain a list of waterbodies within their jurisdictions that fall short of applicable federal water quality standards. In addition, a state must develop a Total Maximum Daily Load (TMDL) analysis for each pollutant. TMDLs define an aggregate level of water pollution for a set of actors connected to a waterway. More specifically, they define the
pollution loading capacities that if met are expected to bring the waterway into compliance with national ambient water quality standards. They then establish the allowable waste load allocations from both point and nonpoint sources. The point sources are then managed through permits under the National Pollutant Discharge Elimination System, while nonpoint sources, which stem primarily from agriculture, are managed through a range of voluntary best management practices.

**NAAQS vs. TMDLs**

Although NAAQS apply to air pollutants and TMDLs apply to water pollutants, they are similar in many ways. As noted, they both involve ambient standards, which define a target of collective performance for a particular group. For NAAQS, the group is defined as the air quality control region, and members of the group are the polluters who discharge the air pollutants within that region. For TMDLs, the group is defined at the level of the TMDL (e.g., the waterway or watershed[s]), and members of the group are the polluters whose activities contribute to the pollution load of that waterbody.

Nonetheless, there are some important differences across these two contexts. A key difference is that 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 apply to the state for failure to meet the ambient standards, and states have regulatory authority for controlling the behavior of group members (e.g., power plant emissions). In this sense, the state can effectively act like a centralized collective for compliance at the air quality control region, setting the rules to control within-group behavior and facilitate compliance with ambient standards. In contrast, for the TMDLs there are no clear penalties for failure to meet the group performance standard and no enforceable rules to control the behavior of all contributing polluters. The CWA does not allow for penalties or regulatory threats, and does not require that TMDLs include reasonable assurances, monitoring plans, or implementation plans. Part of this limitation stems from the fact that the CWA provides no general regulatory structure for limiting nonpoint source pollution. Thus, in the context of TMDLs, the group is more akin to a decentralized collective.

Given these differences, based on the theoretical and empirical literature on collective approaches discussed above, we would expect the NAAQS to be more effective than the TMDLs and that seems to, in fact, be the case. The CAA, along with the NAAQS provision, is generally viewed as having been an effective environmental policy. It has led to significant reductions in all criteria air pollutants over several decades (EPA 2017). In contrast, there is no strong evidence that TMDLs have been broadly
effectiveness in improving water quality. Evaluations have been limited for several reasons, including a shortage of monitoring data and the fact that many TMDLs were developed recently and may take a long period of time to reveal effects (Norton et al. 2007). In many cases, implementation relies on voluntary efforts by farmers, an approach that is unlikely to be effective without strong incentives or requirements to curb polluting activities (Segerson 2013). Thus, when viewed from the perspective of a group-level policy, there are reasons for concern about whether TMDLs will emerge as a useful approach to improving water quality.

Conclusion

The defining feature of a group-level policy is the setting of penalties or rewards based on measures of group performance, or the allocation of rights to a group rather than individuals. While group-level policies are common in environmental and natural resource settings, economists tend to focus attention on policies at the individual rather than group level. The existing research on group level-policies applies to specific areas of environmental and resource management, and this leaves a gap in the literature on cross-cutting themes about the design of these instruments and lessons learned about the successes and failures of their implementation. In order to identify theoretical and empirical insights that will help inform policy design and implementation, along with future research, this paper reviews the way that group-level policies have been applied and studied in a variety of contexts.

The theoretical literature shows that group-level policies can be designed to effectively meet environmental and natural resource management objectives. However, effectively meeting an objective does not necessarily imply efficiency or cost-effectiveness. In many cases, the overall efficiency of group-level policies will depend not only on how the group-level instrument affects individual incentives, but also on the internal operating rules and norms of the group itself. Both the policy-induced incentives and institutional arrangements matter because of the way that certain group-level instruments are susceptible to free-riding and multiple equilibria.

The empirical literature reviewed herein generally reinforces these findings. Much of the early work on group-level policies in an environmental and natural resource context focused on nonpoint source water pollution. The empirical literature in this area relies heavily on laboratory experiments and serves mostly to confirm that group-level instruments affect individual behavior in the ways that theoretical models predict. In contrast, evaluations of PES programs are based on studies in the field. While many of the observational studies face standard challenges of rigorous program evaluation, some
researchers have conducted field experiments. Many of these studies underscore the importance of trust and social capital for the success of group-based approaches. Industry-wide voluntary programs are another area where group-based policies have received attention, and while the evidence is mixed on the overall environmental effects of these programs, the existing research shows the importance of free-riding and selection effects for evaluating effectiveness, efficiency, or both. Finally, the design of many rights-based policies for fisheries management have group-based elements because of the way that penalties or rewards are triggered or allocated. Research in this area based on the evaluation of actual policies once again shows the importance of coordination among group members, often through the existence or formation of fishing cooperatives.

In conclusion, we find that a unified conceptual framework is helpful for looking across group-level policies in different areas of environmental and natural resource management. Although contained in distinct pockets of the literature, the existing research provides broad insights about the design features that affect the potential success and failure of group-level approaches. Identification of these elements can aid in the evaluation of existing policies and guide future design and implementation. For example, our comparison of the NAAQS and TMDLs as part of the CAA and CWA in the United States highlights the importance of having clearly defined penalties and rewards, along with the ability for monitoring and enforcement. While obvious elements for policies that focus on individual action, they are crucial to the design of group-based policies as well. We view the contribution here as an early step toward a more unified perspective on group-level approaches—one that we hope guides future research and brings greater recognition to the existing use and future potential of group-level approaches for environmental and natural resource policy.
References


