PERC POLICY REPORT | FEBRUARY 2019

DISTRIBUTIONAL EFFECTS OF ENVIRONMENTAL MARKETS:

INSIGHTS AND SOLUTIONS FROM ECONOMICS

EDITED BY CHRISTOPHER COSTELLO

PERC—the Property and Environment Research Center—is a nonprofit research institute dedicated to improving environmental quality through markets and property rights. Located in Bozeman, Montana, PERC pioneered the approach known as free market environmentalism. PERC's staff and associated scholars conduct original research that applies market principles to resolving environmental problems.

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Introduction

Christopher Costello (U.C. Santa Barbara, NBER, and PERC)

Perhaps more than at any point in history, governments, conservation groups, and private organizations are harnessing property rights and markets to manage common-property resources such as fish, water, groundwater, biodiversity, and climate. Anecdotes suggests that the allocation of rights or permits, the rules governing ownership and trading, and the longevity and security of rights are paramount to adoption and long-run success. In particular, the distributional consequences of different trading structures can significantly impact the likelihood of adoption of environmental markets, enforcement success, long-term viability of the market, public perception of the market, and even efficiency of outcomes. For example, if too many groups are disadvantaged or excluded from the rights-allocation process, or if all of the economic benefits of improved management are taxed or auctioned away at the outset, then an otherwise successful market approach may become politically unviable.

Despite growing evidence that the distributional consequences of environmental markets can play a central role in their adoption and durability, only a small body of scholarship explicitly tackles these challenges. In July 2018, the Property and Environment Research Center (PERC) hosted a workshop to assemble world-class researchers and practitioners to shed light on the ways in which environmental markets could be better designed and implemented to explicitly account for distributional concerns. To achieve this goal, we attempted to make three contributions: First, we sought to convene leading researchers on the distributional effects of environmental markets. Second, we commissioned a set of short papers and presentations by these experts, with the goal of proposing novel and actionable approaches to account for distributional effects in market design and implementation. Third, we sought to pair each presentation with an applied practitioner with on-the-ground experience who could help hone the idea either into a more applied, policy-relevant research contribution, or, in some cases, even into an actual policy proposal.

What is the justification for better understanding the role market design plays in the distributional consequences of the transition to market-based environmental protection? Our motivation stems from the belief that by better understanding this connection, future environmental markets can be designed for increased uptake and durability, and that this information will likely also lower the cost and ultimate success of implementation.

Workshop Format

The format of the workshop was non-traditional. We commissioned a total of 10 papers from internationally regarded scholars in some aspect of the distributional consequences of environmental markets. These scholars proposed their own paper topics, and workshop organizers helped refine these topics to ensure a well-rounded final set of ideas. The papers were deliberately written to contribute to the format of the workshop—they were short, concise and provocative, and were not simply carved out of existing papers written for a different audience. The papers were both theoretical and empirical. In some cases, they were highly generalized across environmental goods—for example, an examination of the role initial allocation plays in distributional effects. In other cases, papers were focused specifically on one environmental domain, such as fishery or air pollution rights.

Based on the content of each paper, workshop organizers identified a suitable discussant to make detailed comments on the conceptual contribution and applied relevance of the ideas being proposed. These discussants were selected from academia, non-governmental organizations, and government agencies. Following the workshop, presenters polished their written contribution based on the discussion, and discussants produced a short summary of their comments. These contributions are compiled in this volume.

Contributors

Lead authors and discussants were selected from a range of universities, non-governmental organizations, and agencies. Here we briefly summarize the paper topics and contributors' backgrounds.

Spencer Banzhaf is a professor of economics at Georgia State University, research associate at the National Bureau of Economic Research, and a senior fellow at PERC. Spencer's paper concerns how to allocate environmental rights to communities to better account for environmental justice. Spencer's discussant is **Kelly Maguire**, the Chief of the Conservation and Environment Branch of the Economic Research Service at the U.S. Department of Agriculture, who also spent 19 years as an economist at the U.S. Environmental Protection Agency.

Corbett Grainger is a professor of applied economics at the University of Wisconsin, Madison. Corbett's paper examines the strategic siting of air pollution monitors and argues that their biased placement exacerbates environmental justice concerns. Corbett's discussant is **Sam Collie**, a PhD student in the Bren School of Environmental Science & Management at U.C. Santa Barbara and a Sustainable Water Markets fellow.

Kelsey Jack is a professor of environment and development economics at U.C. Santa Barbara and a Faculty Research Fellow at the National Bureau of Economic Research. Kelsey's paper examines whether using auctions to allocate payments for ecosystem services in developing countries systematically disadvantages the poor. Kelsey's discussant is **James Salzman**, a professor of environmental law at U.C. Santa Barbara and UCLA whose scholarship and applied work is in ecosystem service provision.

Kailin Kroetz is a fellow at Resources for the Future (Washington, D.C.) in the Land, Water, and Nature program. Kailin's paper examines whether implementing an environmental market in a fishery (called "catch shares") leads to leakage where fishermen exit that fishery, and enter less-regulated fisheries, thus undoing some of the environmental benefit of the market. Kailin's discussant is **Jono Wilson**, who is a lead scientist on global fisheries for the Nature Conservancy, one of the world's leading conservation-based non-governmental organizations.

Kyle Meng is a professor of environmental economics at U.C. Santa Barbara, a Faculty Research Fellow at the National Bureau of Economic Research, and a former fellow at Environmental Defense Fund. Kyle's contribution focused on the environmental justice effects co-pollutants in California's cap and trade program. Kyle's discussant is **Larry Goulder**, an economics professor from Stanford University and a fellow at the National Bureau of Economic Research.

Dominic Parker is a professor of economics at University of Wisconsin, Madison, and a senior fellow at PERC. His paper examined the distributional effects of conservation easements, focusing on the tax policies that incentivize landowners to donate easements to land trusts. Nick's discussant is **Mike Conner**, the director of land protection in California for the Nature Conservancy.

Kathleen Segerson is a distinguished professor of environmental economics and associate dean at the University of Connecticut. Kathleen's paper examines the distributional and efficiency implications of allocating property rights in environmental markets to *groups* instead of to *individuals*. Kathy's discussant is **Merrick Burden**, a senior economist with the fisheries solution center at the Environmental Defense Fund.

Matthew Zaragoza-Watkins is an assistant professor of energy and environmental economics at Vanderbilt University. Matt's paper concerns the possible distributional effects of leakage that can arise from carbon pricing policies. Matt's discussant is **Daniel Kaffine**, a professor of environmental economics and a Renewable and Sustainable Energy Institute fellow at University of Colorado, Boulder.

Christopher Costello is a professor of resource economics at U.C. Santa Barbara, a research associate at the National Bureau of Economic Research, and a senior fellow at PERC. Chris's paper examines the efficiency and equity consequences of alternative approaches to initially allocating rights in an environmental market. Chris's discussant is **Nicole Sarto**, a senior specialist at the fishery solutions center at Environmental Defense Fund.

Andrew Plantinga is a professor of resource economics at U.C. Santa Barbara. Andrew's paper examines the distributional consequences of using market-based mechanisms to manage spatially connected and spatially heterogeneous natural resources. Andrew's discussant is **Kerry Smith**, a regent's professor of economics and a distinguished sustainability scientist at Arizona State University.

Themes

Each of the paired papers and discussant comments compiled in this volume represents an important advancement in both our academic understanding of the distributional effects of environmental markets and in the practical solutions that might be applied in real-world settings. These 10 distinct contributions fall roughly into three thematic areas. In what follows, I attempt to succinctly highlight the key contributions of each paper within these themes, with a focus on how the paper and discussion contribute to our understanding of distributional effects.

Theme 1: Environmental Justice Concerns and Solutions

That disadvantaged communities tend to experience disproportionate levels of pollution has been well-established in the literature, and it is this finding that underpins the field of environmental justice. But to focus more explicitly on solutions, a more nuanced set of questions may be warranted. These questions concern issues such as: (1) whether the correlation between race or income and pollution is causal, (2) whether regulators focus pollution control toward rich, advantaged communities, (3) whether gentrification is the ultimate outcome of a cleaner environment, and (4) whether institutional solutions exist that empower communities with sovereignty and bargaining power over environmental quality. These are exactly the kinds of questions tackled in several papers in this volume.

Spencer Banzhaf argues that a Coasean approach (after the Nobel Prize-winning economist Ronald Coase) could be taken to help resolve environmental justice concerns between a community and a polluter. He argues that if rights (for example, to a clean environment) are allocated to communities, then any bargaining between the community and the polluter could plausibly lead to an actual Pareto improvement, in which all parties are better off. Under this line of thinking, there is an important role for environmental justice advocates as facilitators of low-transaction-cost bargaining and to ensure that rights (to communities) of a clean environment are not expropriated.

Corbett Grainger examines whether air pollution regulators strategically site pollution monitors in clean locations away from disadvantaged communities. This is difficult to study empirically because it is hard to know the pollution levels in areas without air pollution monitors. Grainger uses satellite data to show that these monitors are placed in atypically clean locations, that this leads counties to be erroneously labeled as "in attainment" with air quality standards, and that this systematically disadvantages poor neighborhoods. He concludes that new sensors and satellite technologies should be adopted to help overcome this strategic bias. Discussant Sam Collie raises the interesting question of whether Grainger's findings are due to strategic placement (of monitors) or avoidance behavior (of people); this remains an open—and crucial—question.

Theme 2: Distributional Effects of Market Mechanisms

A central theme that emerged out of this workshop is that the way we design environmental markets can have significant distributional consequences. While these design features have received some attention because they may also affect efficiency, few scholars have specifically focused attention on the distributional implications of design features such as (1) ownership rights in alternative environmental markets, (2) how trading affects co-pollutants and environmental justice, (3) tax incentives for ecosystem service provision, and (4) conservation obligations across heterogeneous communities.

Kailin Kroetz tackled the complex and difficult-to-study issue of leakage in fishery markets. Because market-based fishery management is usually implemented one fishery at a time, there is a risk that regulated fishermen will sell their grandfathered rights and enter a less well-regulated fishery, possibly undermining the sustainability of the policy and affecting the distributional outcomes in both fisheries. She raises the profile and importance of leakage and, as echoed by discussant Jono Wilson, argues for a more coordinated policy across fisheries.

Environmental markets designed to solve one problem may help solve (or exacerbate) another problem. Kyle Meng examines this kind of "co-benefit" in California's carbon cap-and-trade market. Because local air pollutants are typically co-emitted with carbon, a reduction in carbon emissions may also lead to a reduction in local air pollutants, and would thus improve air quality. Trading of carbon allowances means that some places will increase pollution while others decrease (though there will be a net decrease). Kyle examines whether this spatial reallocation of pollution exacerbates, or ameliorates, environmental justice concerns.

Dominic Parker also examines the distributional consequences of an incentive-based approach to environmental protection, focusing on federal tax incentives to provide conservation easements on land. Easements are voluntarily donated by a landowner, and as a consequence, the land is permanently conserved. In exchange, the landowner obtains generous tax incentives, though these tax incentives primarily extend to wealthy landowners. Parker raises concerns about the quality and additionality of conservation easements because, under current policy, donated land may be of questionable conservation value. On the other hand, his discussant Mike Conner argues that various practical mechanisms are in place to ensure robust conservation benefits from these easements.

While other authors focused on the distributional consequences of environmental policy across income, race, or other classes, Andrew Plantinga focuses on the distributional effects across space, for example across diverse communities. In his spatial model of a renewable resource, Plantinga argues that one owner's resource extraction affects all other owners for mobile resources such as water, fish, and game. The common-pool nature of this problem leads to over-exploitation of such spatial resources, and Plantinga proposes a new market-based instrument, based on transferable conservation obligations, to overcome this challenge. Discussant Kerry Smith helps bridge this idea with important insights from the public economics literature.

Theme 3: Allocation of Rights and Responsibilities in Environmental Markets

Perhaps the most fundamental design feature of any environmental market is the initial allocation of rights and responsibilities to different actors. In practice, most environmental markets grandfather rights to historical users, and several authors comment that this can have important distributional, and even efficiency, consequences, though it is rarely studied. In this theme, authors tackle various challenges around the allocation of rights, and their research gives rise to several important new insights that could potentially improve buy-in, equity, durability, and efficiency of new environmental markets.

We often take for granted that rights will be allocated to an *individual*, such as a fisherman (for fish), a polluting facility (for air pollutants), or a farmer (for water). Kathleen Segerson challenges this assumption, and instead asks whether allocating to *groups* can bring previously unrecognized benefits. A group allocation of rights links members' actions, and it has both distributional and incentive consequences. In this setting, the payoff is typically tied to group performance, rather than individual performance, which brings about free-riding incentives, social pressures, risk sharing, and other important features that are not obviously present when allocating to individuals. Discussant Merrick Burden emphasizes that group allocations can be contentious because members must agree on how to share benefits.

My own contribution argues that the initial allocation of rights can have important political economy consequences. If all rights are grandfathered for free, then incumbent resource harvesters are typically made better-off under the market than under a status-quo management system. This incentivizes them to endorse the market approach. But if rights are auctioned, are too costly, or if previously well-off users received small allocations, then it is likely that some incumbents will oppose the market. Other authors have argued that free allocations represent a windfall benefit, often to those who historically created environmental problems. I show that alternative approaches to allocation, based on a concept of “merit,” can both reward historically responsible actors and sufficiently reward incumbent resource users such that a broad set of stakeholders endorse the transition to an efficiency-enhancing market.

When an environmental market covers only part of an economic sector (for example, when one country unilaterally adopts a carbon cap-and-trade program), there is a risk of leakage—where price effects cause pollution to decrease where the market is implemented but increase in other locales. Matthew Zaragoza-Watkins examines leakage of this form for California’s carbon cap-and-trade program. To combat leakage there, an output-based subsidy is applied in the form of a free allocation of rights, which incentivizes increased output. This output-based subsidy dampens, or eliminates, the price response that would cause leakage. He finds that indeed emissions in California have decreased, but output has remained roughly constant, as a consequence of this policy.

In some cases, economic incentives involve direct payments for ecosystem services, where a landowner, fisherman, or other agent is paid to voluntarily provide an environmental benefit. To avoid the problem of additionality (where a large payment is made for a conservation action that would have occurred even without the payment), and thus incentivize truthful bidding, auctions are often used. Kelsey Jack asks whether these auctions disproportionately favor the rich—after all, the rich may be in a better position to place high bids. She finds just the opposite. Using a randomized field experiment in Malawi, she finds that auctions typically favor worse-off households.

Conclusions

This workshop has shown that as communities of people around the world increasingly run up against natural-resource constraints, environmental markets can be thoughtfully designed to deliberately account for distributional equity while simultaneously accounting for environmental protection and economic efficiency. Achieving these multiple objectives is possible because there are so many dimensions to the design of an environmental market, such as allocation of rights, to whom they are granted, the durability of rights, how environmental quality is monitored, and how co-benefits or co-costs are considered. A corollary is that the distributional consequences of a market may actually affect its efficiency because market approaches that are deemed unjust may fail to be adopted in the first place. This body of work represents a small, yet we hope provocative and informative, first step toward a much larger research agenda addressing the distributional effects of environmental market design.

1 Environmental Justice and Coasean Bargaining

H. Spencer Banzhaf (Georgia State University, NBER, and PERC), Lala Ma (University of Kentucky), and Christopher Timmins (Duke University and NBER)[1]

Environmental markets are sometimes criticized for reducing regulatory control over the distribution of environmental improvements and potentially increasing disparities in the exposure to pollution (Chinn 1999). The correlation between pollution and the presence of poor and/or minority households, well established in the environmental justice literature, thus raises a possible tension between the efficiency gains of markets and desired distributional effects.[2]

Large-scale markets like cap and trade mainly involve exchanges among polluters. Unless firms with higher abatement costs (and hence higher demand for pollution permits) systematically lie upwind of environmental justice communities, there is no a priori reason why we would expect pollution trading to worsen such correlations. Perhaps for this reason, the empirical literature has not found any clear consensus about how permit trading affects the distribution of pollution (Bento, Friedman, and Lang 2015, Fowlie, Holland, and Mansur 2012, Grainger and Ruangmas 2017, Shadbegian, Gray, and Morgan 2007, and Meng, this volume, ch. 4).

A related concern, however, is the distributional effect of trading pollution rights between the "green" and "brown" sides of the market, as with Coasean bargaining between polluting firms and local communities. Here there is, a priori, a more plausible reason to expect that trading would steer pollution to poorer neighborhoods, if those neighborhoods are more willing to accept lower compensation for pollution, have weaker bargaining power, or are less informed about the costs of pollution.

Although Coasean bargaining may pose distributive challenges, a Coasean perspective also provides new ways to think about old environmental justice problems. For example, previous work has emphasized the possibility that, absent Coasean-style transactions, cleaning up the environment, even in poor areas, can have unintended distributional effects if it destroys jobs or increases property values. In the latter case, the problem is that when local environments improve, richer households may move in, bidding up rents to their own higher willingness to pay, forcing the poor to pay more for the improvement than they are willing (or able) to pay, a process sometimes called "environmental gentrification" (Banzhaf and McCormick 2012).

Is there a way to overcome this problem? To turn situations, whether involving polluting activities or clean-up activities, where the gains are great enough that everybody *could* be made better off (so-called "potential Pareto improvements") into situations where everybody actually gains (or "strict Pareto improvements")? To do so, it would seem that some kind of right, other than property rights in land, must be held by local residents (including renters). In the long run, this may be difficult: If renters anticipate the rewarding of such rights, their value may be dissipated by activities to capture them, or simply capitalized themselves into land values (Anderson 2012). Here, we take a more short-term approach, or consider one-off events that are not anticipated.

The Coase Theorem

Coase's famous theorem is a starting point for understanding such trades and negotiations. The Coase theorem states that, under well-defined property rights and in the absence of transaction costs, it does not matter who holds the property rights to the use of the environment because negotiation and market transactions will ensure the same, efficient use of resources (Coase 1960). Through negotiation, the right to pollute (or to be spared pollution) will end up in the hands of the individuals or firms who value it most, and all parties will be appropriately compensated for any nuisance or foregone profits they consequently bear. However, the distribution of wealth will depend on the initial allocation of property rights.

Firms have preferences over where to locate their industrial facilities and a willingness to pay to locate at a certain place (e.g., Wolverton 2009). At the same time, households have a tolerance for pollution and some willingness to accept compensation for industrial activity nearby, which might lead poorer households to "come to the nuisance" (e.g., Banzhaf and McCormick 2012, Been 1994). While these two mechanisms, taken in isolation, have been discussed at length in the environmental justice literature, a Coasean perspective triangulates them, seeing the potential for transactions between the two sides: Polluting facilities will go to areas that have the highest value to them, net of the compensatory demands of local residents.

Consider a facility that emits pollution into the surrounding environment at zero cost to itself. Each additional unit of pollution emitted creates a benefit to the facility in terms of not having to use an expensive abatement technology or not having to forego production. These marginal benefits to the facility from emitting pollution, *MB(E)*, are declining until a point where they reach zero, as depicted in Figure 1 at E_0. With no private costs of generating pollution, the facility would choose to generate pollution up to E_0. But pollution also affects nearby households in the form of increased health risks, unpleasant odors, and other disamenities. For simplicity, we assume that each additional unit of emissions inflicts a constant marginal external cost on local residents, *MEC(E)*.

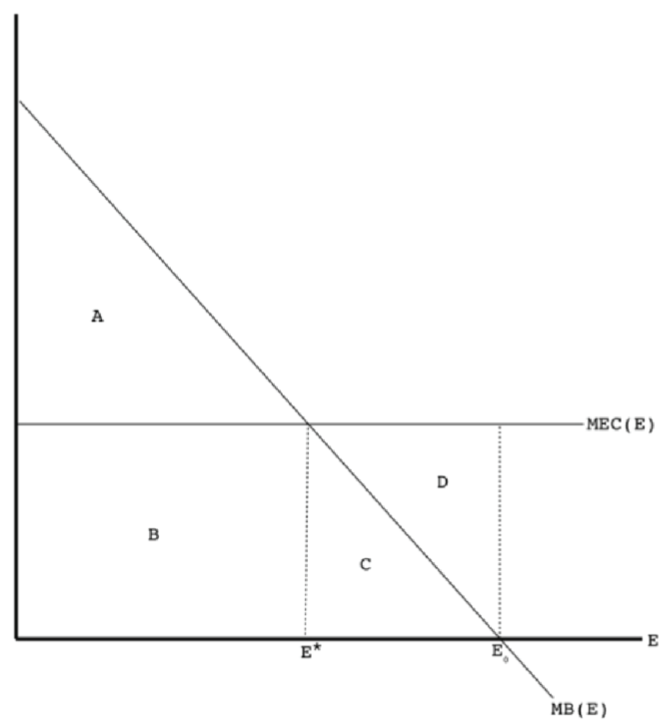
Suppose further that local residents hold the relevant property rights, in the sense that they can veto polluting activities or accept them. Even if legislation does not explicitly recognize this right, residents may be able to assert it through tort law, zoning laws, holding up permitting processes, political protest, and so forth. In the absence of any sort of compensation, local residents would prefer that the facility release no emissions. From a Coasean perspective, however, they would be willing to accept payments greater than or equal to *MEC(E)* per unit of pollution. For example, for E^* units they would accept payments greater than or equal to area B in Figure 1. Meanwhile, the polluter will be willing to pay the residents up to $A+B$ for the right to generate E^*. This creates the opportunity for trade, maximizing social surplus at E^*, with a division of the Coasean surplus (area A) in some fashion between the two parties, according to their relative bargaining power. Such payments may be cash transfers, or they may take the form of local jobs, investments in parks and community centers, and so forth (Jenkins, Maguire, and Morgan 2004).

Note that polluters have an incentive to locate where *MEC* (and, therefore, willingness-to-accept compensation of local residents) is low, so as to lower their required compensation payments (area B). A lower *MEC* is also compatible with a higher efficient level of pollution. *MEC* might be low for two reasons. First, the location may be remote, with fewer people, so that total injuries are lower. Second, local residents' willingness-to-accept compensation for injuries may be lower. In either case, Coasean bargaining theory would treat this incentive as one leading to economic efficiency. However, this latter incentive might also give rise to environmental justice correlations (Hamilton 1995). It is likely to be poorer or minority households who have a lower willingness to accept compensation, perhaps because they may prioritize the consumption of other important goods (even basic needs like food and shelter) over the environment.

The model of Coasean bargaining suggests that the observed distribution of pollution is efficient—*given the distribution of income*. This logic tends to shift the locus of injustice back to inequality in the underlying distribution of income and wealth. However, in the Coasean case, the environmental resource is a valuable asset, and its allocation itself represents part of the distribution of wealth. Environmental justice concerns—and distributional concerns generally—would suggest allocating the property right to communities.

If allocated those rights, local communities could then keep the right to be free of polluting facilities, or negotiate with polluters, as they saw fit. If they have full information and full power to bargain (admittedly big "ifs"), they cannot be worse off if they allow polluting facilities to operate. That is, in the Coasean case, such pollution not only would be efficient in the sense of a potential Pareto improvement, but in the sense of a strict Pareto improvement. Both sides are made better off by the exchange. This observation highlights one potential

Figure 1. Coasean Bargaining



area of tension in the understanding of environmental justice. While in many cases a focus on equity of exposure to pollution (a kind of distributive justice) and a focus on the ability to participate in decision-making (procedural justice) run together, they do not always. Justice entails sovereignty over environmental decisions, and thus the right to accept polluting firms as well as to reject them (Foster 2001).

There is some limited evidence of Coasean logic at work. For example, Coasean dynamics appear to take place in the form of host fees collected by neighborhoods near landfills. Jenkins, Maguire, and Morgan (2004) find that citizen participation in host fee negotiations leads to greater host compensation. Similarly, Hamilton (1995) finds that communities with lower voter turnout were more likely to see local firms expand their processing of hazardous wastes.

A Property Rights Perspective

Moving beyond the Coase theorem as a special case of Coase's analytical apparatus, we can think about the distribution of property rights, trade in those rights, and transactions costs and other barriers to trade as key parameters for understanding the distributional issues underlying environmental justice concerns.

If property rights are ambiguous rather than well defined, local communities may undertake activity to claim de facto rights. For example, they might exploit zoning and permitting processes to block pollution, activities that can be interpreted as the claiming of rights or at least power. However, environmental justice communities may have more difficulty asserting or claiming such rights through political action (Hamilton 1995). Less access to the corridors of power, less formal education and legal expertise, language barriers, and other such disadvantages may limit their understanding of what is possible or their ability to exploit it. If they live in areas with numerous polluters, they also may face the problem of having to negotiate on several fronts at once to address cumulative impacts.

Suppose communities do have or obtain at least partial property rights to the local environment. Suppose further that some transaction or change is being contemplated to reduce pollution. As noted previously, one distributional consideration in this setting is that gentrification effects following the improvement might harm renters, especially the poor. Giving these groups a say in the negotiations can help alleviate such concerns. At one extreme, they might veto the change. More likely, they might negotiate changes in the community that are more likely to appeal to existing residents rather than gentrifiers. Alternatively, they might negotiate low-income housing. Low-income housing, often criticized by economists as an inefficient way to redistribute overall wealth, might be viewed differently when part of a package of local policy options leading to actual Pareto improvements.

Suppose now instead that some transaction or change is contemplated that would worsen environmental quality, such as the siting of a polluting factory or waste facility. Firms looking to site such activities will search for the area with the lowest willingness to accept compensation. As noted previously, such areas are likely to be in poorer communities. Moreover, because of the way they are disadvantaged, environmental justice communities may not be able to negotiate as well as the firms they are negotiating with, nor as well as other communities where the firm might consider moving. They face the same issues when negotiating as when asserting the right to the environment in the first place. They also may have more difficulty overcoming the free-rider problem on their side of the negotiation: Whereas the benefits of polluting are a concentrated interest for one firm, the costs of polluting are dispersed among residents (Yang and Kaffine 2016). As a consequence, environmental justice communities, when in negotiations, may end up systematically with a small share of the Coasean surplus (Anderson 2012). If they are uninformed about the full extent or harm of pollution, they might even obtain less than their damages (area *B*), acting as if *MEC(E)* were lower than its true value. Reasoning back from such outcomes, this would further undermine their incentive for overcoming the obstacles to assert property rights in the first place. It also would further push firms to those areas.

Recently, Vissing and Timmins (2017) have explored information asymmetries in Coasean bargaining. They examine the contents of leases signed between shale gas operators and households in Tarrant County, Texas, for the rights to extract natural gas. The terms of these leases dictate both payments in the form of royalty compensation and protective clauses designed to reduce health and environmental risks from the extraction process. After conditioning on income, Vissing and Timmins find that black homeowners are less likely than whites to have any of the protective clauses, and are more likely to receive a smaller lease payment. Hispanics, interestingly, are as likely as non-Hispanic whites to have protective clauses and a similarly high royalty payment *unless* they reside in a census tract where households tend to be linguistically isolated—the latter suggesting a form of information asymmetry in the bargaining process.

In addition to negotiating rights with polluters, environmental justice communities often must negotiate with other members of the local community. More generally, we might ask what it even means to allocate rights to local communities. What is the community, and who speaks for it? In some cases, community organizations may assert rights, in the ways discussed above. In other cases, it may be a case of a local government, such as a county, acting on behalf of its citizens. While local governments are more likely to understand and represent local interests than higher levels of government, these scenarios raise potential agency problems, in which those bargaining on behalf of victims are not actually those bearing the costs of the pollution. The marginal cost to victims of another unit of pollution may actually be $MEC(E)$, but the cost internalized by the agents representing them may be something lower, leading to more pollution and less compensation.

Such themes are well illustrated by the cases of Warren County, North Carolina, and Kettleman City, California, where environmental justice communities felt misrepresented by their local governments in negotiations with polluters. In Kettleman City, described in Cole and Foster (2001), a waste management firm proposed building a toxic waste incinerator at a nearby dump site. Located in the San Joaquin Valley, Kettleman City was 90 percent Hispanic, with 40 percent of residents speaking no English. Through inadequate provision of public notice, the begrudging provision of translators at public meetings, and the scheduling of those meetings in difficult-to-reach locations, it was clear that information asymmetries were part of a strategy to inhibit local participation. Despite vigorous protests from Kettleman City residents, Kings County initially approved the deal. The county was to receive $7 million annually from the deal, but these benefits were spread over a 1,400-square-mile rural county, with a demographic very different from Kettleman City, while the environmental injuries were concentrated in the city.

The Kettleman City example demonstrates how political economy problems can overcome Coasean forces even at the county level. Such forces may be even stronger at the state level. The Cerrell Report (Cerrell Associates 1984) provides an infamous example of an effort to direct the siting of polluting facilities toward communities that would be ineffective bargainers. A consulting report requested by the California Waste Management Board, a state agency, the report identified characteristics of local communities that would not protest the location of waste sites in their area—in particular, people who were without a college education, had low incomes, were Catholic, and were "not involved in voluntary associations." Additionally, one might view the more recent case of Flint, Michigan, as a breakdown of efficient Coasean bargaining. Local preferences were ignored in what were essentially negotiations over the local water source and resulting pollution and were replaced with the preferences of Michigan's Treasury Department.

Concluding Thoughts

The role played by environmental justice advocacy groups, both descriptively and prescriptively, can be understood in part through a Coasean lens. Such groups can play the role of monitors, looking for places where rights to a clean environment are being expropriated. They can provide information to environmental justice communities, helping them search their own preferences for the level of external pollution costs, MEC in our stylized diagram. And they can provide legal and other services in negotiations, lowering transactions costs. Such activities can help make Coasean markets fair as well as efficient.

Endnotes

1 We thank Kelly Maguire and other participants in the PERC workshop for comments. Portions of this paper, including Figure 1 and its discussion, are adapted from Banzhaf, Ma, and Timmins (2018).

2 Overviews can be found in Banzhaf (2012), Banzhaf, Ma, and Timmins (2018), Bullard (1994), Cole and Foster (2001), and Noonan (2008).

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RESPONSE

Kelly Maguire (USDA, Economic Research Service)

The mechanisms that give rise to concerns about the distribution of environmental outcomes, particularly as they relate to low-income and minority populations, are complex. While environmental markets offer an efficient solution to reducing negative externalities, they may create or exacerbate distributional outcomes related to pollution. The environmental justice concept was born out of concern over the unequal distribution of hazardous waste landfills in Warren County, North Carolina, when local residents questioned why such facilities were predominantly located near poor, black communities (UCC 1987). Since that time, economists and other social scientists have worked to understand the factors that give rise to such disparities and how market solutions could result in more equitable outcomes.

The Coasean bargaining model offers a conceptual method for thinking about distributional issues. As discussed by Spencer Banzhaf, in the presence of well-defined property rights and the absence of transaction costs, those who are willing to pay to pollute will negotiate with those who are willing to accept pollution to result in an efficient allocation of the resource. If the market works, then a Coasean solution or outcome should arise, and all parties should be satisfied with the outcome, even though some would be subjected to more pollution than others.

Why then do we see concerns about the distribution of pollution? Banzhaf offers several explanations for how Coasean bargaining may not result as intended and environmental justice concerns arise. Local communities may in fact lack property rights over the resource, or perhaps those rights are ambiguous or ill defined. Even if property rights are well defined, communities may lack the ability to negotiate with more powerful firms.

In addition to concerns about the initial allocation of property rights, two other features of the Coasean bargaining model could give rise to distributional concerns. Transaction costs, as also discussed by Banzhaf, may be high—often very high. Low-income communities may simply lack the resources to communicate with residents or organize meetings and events to determine how to negotiate with a polluter, or they may even lack awareness of the implications of the presence of pollution. These costs may be insurmountable for a community struggling to meet basic needs like clean drinking water or a sustainable food supply.

Second, the Coasean bargaining solution relies on a limited number of participants in the negotiation. Many communities concerned about environmental justice are dealing with not just one pollution source but multiple sources for which the cumulative impacts may be great and unknown. In addition to a hazardous waste landfill, a community may be located near a major transportation corridor, in close proximity to an industrial park, and have drinking water issues. Any one of these pollutants could be of concern, but when taken together the synergistic impacts are unknown, and the number of parties with whom the community would need to negotiate is daunting, making negotiation infeasible.

While the Coasean solution offers a conceptual model for considering how environmental justice concerns may arise or be ameliorated, many of the foundational components make it unlikely to expect that the market will efficiently and equitably allocate the distribution of pollution between those who are willing to pay and those who are willing to accept negative externalities. Banzhaf suggests a role for environmental-justice advocates to augment the features of the Coasean model that could result in a more equitable outcome for communities with distributional concerns by identifying property rights and working to reduce transaction costs. These options would certainly go far in framing a market solution to distributional considerations.

Alternatively, or perhaps in addition to Banzhaf's suggestions, it may be the case that communities with environmental justice concerns would benefit from efforts to provide a baseline level of environmental quality that is greater than what they currently experience. Provision of a similar level of environmental quality for all communities, regardless of race, ethnicity, or income and for all negative externalities could alleviate some distributional concerns. Any deviations above the baseline could then be negotiated through a Coasean bargaining solution.

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2 Environmental Justice and Ambient Air Pollution Monitoring

Corbett A. Grainger (University of Wisconsin, Madison)

There are many potential channels through which environmental regulations can have distributional implications, and economists are increasingly focusing on disentangling the impacts of environmental policies on different populations (Fullerton 2011). Identifying these impacts is challenging because environmental policy has widespread impacts, including changing the provision of environmental amenities like air quality; affecting returns to capital, wage rates, and the costs of other inputs for firms; impacting the prices of intermediate and final consumer goods; and affecting house prices and land rents as individuals move according to their demand for environmental goods.

While the previous literature has documented efficiency and distributional impacts of air pollution regulations, little attention has been paid to how ambient air pollution is actually measured. In many countries, including the United States, ambient air quality standards are set by a central authority (for example, the U.S. Environmental Protection Agency). Under the Clean Air Act and its amendments, the federal government sets National Ambient Air Quality Standards and delegates authority to local authorities (state, tribal, or substate agencies) to monitor ambient air pollution and report the results to the EPA. The siting decision for in situ pollution monitors is also delegated to the local regulator. Because the local regulator faces potential penalties for exceeding the pollution standard, there is an incentive to avoid polluted areas when siting a new ambient pollution monitor. That is, holding all else equal, the local regulator has an incentive to site new monitors in areas that are less likely to trigger nonattainment designations by the EPA.[1]

Grainger et al. (2018) employs satellite-derived pollution estimates to show that local regulators strategically site new pollution monitors in areas that are relatively clean compared to the surrounding area. I argue that this is important for both efficiency and distributional reasons. If air pollution standards are set uniformly (such as with the National Ambient Air Quality Standards) but enforcement is heterogeneous due to strategic monitoring of ambient air pollution, the regulations may be less efficient than previously believed. Moreover, because pollution hotspots effectively go unnoticed by the regulator, there could be important distributional implications.

One could imagine two types of distributional impacts of ambient air pollution monitoring. First, it has been widely documented that poor and minority households tend to live in areas with higher levels of ambient air pollution (Banzhaf 2012). This is largely due to Tiebout sorting in housing markets: As air quality in one area improves relative to other areas, higher-income households tend to "bid up" housing prices in that neighborhood, given that willingness to pay for environmental amenities is an increasing function of income. A large literature in economics highlights the impact of air quality regulations on housing prices, demonstrating a robust negative relationship between pollution and housing prices (e.g., Timmins et al. 2015). My ongoing research finds that local regulators avoid the most polluted places at the margin, which would disproportionately affect poor and minority communities. Indeed, our estimates suggest that many areas of the country would be in violation of the National Ambient Air Quality Standards, but because these areas are not monitored, those areas remain classified as being in attainment with federal regulations. However, whether or not this has a net positive or negative impact on any given community is a nuanced question, as it depends on individuals' marginal willingness to pay for air quality (see Banzhaf et al., this volume, ch. 1).

A second distributional impact of ambient air pollution monitoring is due to political economy considerations. The local regulator, when siting a new ambient air pollution monitor, faces a complicated decision. The regulator has limited resources for monitoring air pollution, and there are many potential reasons for siting a monitor in any given location. For example, pollution monitors are used to protect human health by detecting elevated levels of pollution; monitors are used for compliance with local and federal regulations; and monitoring data are used for modeling by scientists and in public health and economic studies. Importantly, though, local regulators may be susceptible to political pressure from influential individuals or firms. Our results suggest that political economy considerations are important drivers of pollution monitoring decisions.

In particular, we find that new monitors are placed in areas that are, on average, relatively clean compared to the surrounding area. Conditional on pollution levels, new monitors also tend to be placed in areas with lower incomes. Furthermore, there is evidence that the interaction of income and pollution matters when considering the siting decision for new ambient pollution monitors: New monitors tend to be sited in areas that are both relatively clean and where incomes are higher. While we cannot identify the underlying reason for this behavior, it deserves further attention by researchers. One obvious possibility, though, is that regulators are simply responding to the most influential voices in society.

This is a new area of inquiry, and more research should be conducted before any broad policy conclusions can be drawn. Our results, however, highlight a few important areas for air pollution policy. First, allowing local regulators to site monitors that determine compliance with federal regulations has led to a classical "principal-agent" problem, whereby the local regulator is responding to perverse incentives by siting new pollution monitors in relatively clean areas. Second, ambient air pollution in some areas (which tend to have low median incomes) likely exceeds the federal threshold, but the areas are classified as being in attainment because monitors have been strategically sited.

What can be done to improve monitoring and realign incentives? Under the Clean Air Act, states will likely maintain their own monitoring programs, and in situ monitoring data will remain the gold standard for compliance. However, remote sensing estimates of local pollution or estimates from mobile monitors could be used to determine a subset of the region where a monitor ought to be placed. This would remove some of the ability of local regulators to strategically site monitors, and it would improve the monitoring network's capacity to detect pollution, target regulatory pressure, and protect human health.

Endnotes

1 As discussed in Grainger et al. (2018), a regulator in a nonattainment county faces a similar problem, but she may face additional federal oversight in the siting decision (such as through the State Implementation Plan process).

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RESPONSE

Sam Collie (U.S. Santa Barbara, Sustainable Water Markets Fellow)

Corbett A. Grainger provides evidence that local regulators engage in careful subterfuge to avoid costly "nonattainment status" of nationally mandated air quality standards. The paper indicates that local regulators, when tasked with siting locations for new air quality monitors, strategically choose locations that (1) have relatively clean air in comparison to surrounding areas, (2) have residents with higher incomes, and (3) conditional on ambient air pollution, avoid poor and minority communities. These findings imply that in many counties, aggregate air quality is worse than nationally mandated standards, yet they avoid costly compliance. In addition, strategic monitor siting allows pollution hotspots to go undetected, meaning that spatial heterogeneity of air quality is more extreme than it would be if siting decisions were made by simply throwing darts at a map.

Two potential mechanisms may be driving these findings. The first is the one discussed in the paper: that local regulators strategically place monitor locations and subsequently retrench from their subversive tendencies. A second possible mechanism at play is that local regulators and/or firms strategically manipulate the spatial distribution of pollution sources in relation to the location of existing air quality monitoring stations. In what follows, I will refer to these as the strategic placement and the spatial avoidance mechanisms, respectively. This tale of two mechanisms has implications for the distributional concerns raised by Grainger.

The first distributional concern is the implication of strategic monitoring on Tiebout sorting, in which individuals "vote with their feet" to reside in locales with environmental amenities that match their willingness to pay for them. Consider the alternate mechanisms in turn. By the strategic placement mechanism, new monitor locations are sited in places that are already dirty; thus, the status quo of ambient air quality is preserved. In this case, siting a new monitor would not spur Tiebout sorting because the spatial distribution of air quality does not change. However, if the spatial avoidance mechanism is at play, then the location of a new air quality monitor would become cleaner after the monitor is commissioned. The change in local air pollution would perturb the equilibrium in the housing market, and Tiebout sorting would occur as a new equilibrium is reached in the housing market.

The second avenue of distributional concerns are principal-agent dynamics of regulatory capture. The hypothesis here is that wealthy individuals have more sway over local regulators than other groups and adopt a "Not in My Backyard" attitude toward air pollution. Once again, consider the alternate mechanisms in turn. The strategic placement mechanism suggests that the principals are the homeowners and their agent is the local regulator. Advocating for a nearby air quality monitor ensures that air quality standards will be enforced locally. If costly nonattainment occurs, the costs are dispersed throughout the county, but the benefits are greatest nearby to the monitor location. The spatial avoidance mechanism, however, suggests that the principal is the regulator and the agents are the polluting firms. Under this dynamic, the regulator seeks to "catch" polluting firms, while the firms seek to avoid detection by altering the location of their emissions.

Empirically testing which mechanism explains why air pollution monitors are in relatively clean places compared to their nearby surroundings is the first step toward deciphering the distributional implications of strategic air quality monitoring. The question remains whether the siting of new air pollution monitors preserves or perturbs the status quo of ambient air pollution and the housing market. Finally, a better understanding of who the local regulators are, and what their motivations are, would shed light on the principal-agent problem of air quality regulation. After all, Grainger has identified that a problem exists in where air pollution is monitored. Coming up with a solution requires an understanding of the mechanism that caused the problem in the first place.

3 Fishery Catch Shares and Leakage of Impacts

Kailin Kroetz (Resources for the Future)

Since their invention in the 1970s, catch shares have become an important tool in wild-caught fisheries management. Catch shares work by establishing property rights, setting a cap on the annual harvest, or "total allowable catch," and allocating shares to fishery participants. There are over 150 catch shares globally representing some of the world's most valuable commercial fisheries; about two dozen are in the United States.

Catch shares can improve both economic and ecological outcomes. One of the main benefits is improved economic efficiency (Homans and Wilen 2005). Increasing profit from the same level of harvest improves outcomes for fishers as well as the communities they support. Catch shares can also have many other ecological and safety benefits. For example, a slower pace of fishing can reduce bycatch—non-target species that are caught and discarded in pursuit of a target species. By allocating rights to a predetermined amount to harvest, safety of fishers can be improved by reducing pressure to fish in poor conditions (Pfeiffer and Gratz 2016). Catch shares can also improve the sustainability of fish stocks through the designation and enforcement of a hard cap on the amount of fish allowed to be caught—the total allowable catch—at a level that current ocean conditions can support (Costello et al. 2008).

Impacts of catch shares can, however, extend beyond the fishery targeted by the policy.[1] Catch shares can change the incentives and constraints facing fishers and prompt them to take individual actions that can affect other fisheries in a region or ecosystem. Responding to the implementation of or changes to a catch share, some fishers may choose to exit the target fishery and region altogether; this is consistent with single fishery management policy and the story that catch shares address the problem of fishery overcapitalization. However, implementation of catch shares may also spur fishers to take actions that are not consistent with single fishery management policy design and evaluation—for example, entering new fisheries, exiting non-target fisheries, and changing catch levels in non-target fisheries. Spillover can occur if the chain of responses by fishers in response to a catch share in the target fishery results in changes in actions in non-target fisheries.

An analogy can be made to leakage in a tradable permits scheme for pollution, where instead of fisheries the unit of analysis is the jurisdiction of the tradable permit program (e.g., region or country). When one jurisdiction comes under a tradable permits scheme, regulated entities may move their activities to another, often unregulated, jurisdiction. This "leakage" of activity outside of the regulated jurisdiction degrades the overall effectiveness of the policy as well as the environmental outcomes in the unregulated jurisdiction.

Spillover in the fisheries context can generate positive or negative impacts as fishers respond to catch shares with actions that meet their economic or other goals. A positive outcome resulting from spillover could occur if fishers adjust their effort to maintain or increase the diversification of their portfolios (i.e., the suite of fisheries in which a fisher participates). Portfolio diversification has been associated with a more reliable flow of income and is said to improve fisher and community resilience (Kasperski and Holland 2013, Sethi et al. 2014, Anderson et al. 2017, Cline et al. 2017). Potential negative spillover outcomes include increased participation and total effort in non-target fisheries, which could result in losses of economic efficiency and decreases in stock levels if the total allowable catches are not well enforced.

Leakage of catch share impacts into non-target fisheries can include changes in several types of distributional outcomes. The most obvious risk is that undesirable fishery outcomes like overcapitalization or low economic efficiency could be shifted from one fishery—the one with the catch share—to another. There are societal risks as well. Depending on the composition of target and non-target fisheries, small-scale fishers could bear the impacts of the policy, which could include exiting fishing or decreased profit. Such outcomes could be likely if the non-target fisheries are small or are commonly fished by smaller operations.

Shifting of effort between fisheries can also affect the resilience of fishing communities and regional fishery systems as a whole. The system-scale perspective is a fundamental attribute of ecosystem-based fishery management (see e.g., Pikitch et al. 2004), a primary of goal of which is to ensure managers account for the complex interactions between the natural and human aspects of an ecosystem in order to preserve its health and resilience. Despite a recognized need, socioeconomic concepts have yet to be operationalized into such ecosystem-based frameworks (Marshall et al. 2017). Spillover of economic activity in response to management change is a critical concept to consider in the context of robust and resilient policy design.

Single-fishery evaluation, whether before or after program implementation, can overlook important consequences of management policies if spillover occurs. Specifically, managers may not incorporate changes in outcomes for all fishers and communities whose livelihoods may be affected by a policy change. Managers could thus be making inaccurate predictions of post-management change by failing to account for the full policy impact. This means we could be learning the wrong—or, at the very least, incomplete—lessons from catch shares, particularly related to socioeconomic and equity considerations. Formally, failing to take into account spillover can result in: (1) inaccurate predictions of post-management changes; (2) failure to evaluate the full impact of a policy; and (3) biased estimates of impact in the target fishery if using other fisheries as controls.

Although spillover is possible through the fisher decision channels described above, there is little empirical evidence of spillover (e.g., Cunningham et al. 2016) and none known to empirically estimate the scope of spillover. Instead, with few exceptions, most economic modeling and evaluation of fisheries policy remains single-fishery focused. Therefore, work is needed to understand the economic connectivity of fisheries, the potential for cross-fishery spillovers, and implications for designing and evaluating catch shares and advancing ecosystem-based fisheries management policies.

Endnotes

1 The target fishery under the management of a catch share system is generally a single species in a single location.

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RESPONSE

Jono Wilson (The Nature Conservancy)

Catch shares are a form of market-based regulation in which fishing participants receive fixed proportions of an annual total allowable catch. Catch shares have stimulated intense debate regarding the social, ecological, and economic implications of privatizing a public good. Reviews of catch shares have generally found that they contribute to reducing the probability of stock collapse (Costello et al. 2008), reduce the variability in catch-quota metrics (Melnychuk et al. 2012), and improve economic efficiencies in the target fishery. The ecological impacts of catch shares were reviewed by Branch et al. (2009), who found mostly positive effects on target species and mixed or uncertain impacts on non-target species and the environment. On the other hand, negative impacts of catch shares are mostly attributed to fairness and equity concerns (Dewees 1998, Macinko and Bromley 2002). Few studies, however, have examined aggregate-level impacts of catch shares across the fishery ecosystem resulting from spillover of effort into non-target fisheries.

As more and more fisheries transition to catch share systems, understanding the positive and negative impacts of catch shares on the greater ecosystem is of tremendous interest to the fisheries management community. Kailin

Kroetz provides a clear rationale for the need to understand spillover effects of catch share systems beyond the target species. Specifically, Kroetz suggests that ignoring aggregate spillover effects from implementation of catch shares may fail to adequately address real impacts to communities, fisheries, and ecosystems, potentially biasing catch share policy debates. Furthermore, Kroetz argues that understanding aggregate spillover effects from catch shares is a prerequisite to achieving ecosystem-based fisheries management.

Distributional effects of catch shares across fishery sectors and the broader ecosystem can have important policy implications (Cunningham et al. 2015). Following the logic put forward by Kroetz, we suggest future work should attempt to understand whether spillover effects lead to net positive or negative efficiency gains across the fishery ecosystem. When considering the impacts of spillovers, the question arises of how to address efficiency changes within the non-target fisheries relative to efficiency changes across the aggregate fishery sector, including both target and non-target fisheries. This distinction will influence the design of coordinated policies across fisheries sectors, including determination of a need for enhanced effort or catch restrictions (a.k.a. sideboards) for vessels operating in non-target fisheries (Cunningham et al. 2015).

Consideration of a coordinated policy across fishery sectors falls within the scope of ecosystem-based fisheries management. Ecosystem-based fisheries management is a holistic approach to fisheries management that recognizes the physical, biological, economic, and social complexities of managing fishery resources (Patrick and Link 2015). In the United States, Fisheries Ecosystem Plans translate ecosystem-level linkages within and across jurisdictions into management guidance (Levin et al. 2018). These plans have been developed in several U.S. fisheries but have yet to demonstrate clear outcomes due to a lack of specific management actions contained within. Policies that mitigate impacts from catch-share spillover may provide that direct link between ecosystem understanding and tangible management actions.

Consideration of catch share systems within an ecosystem-based management context is paramount as the effects of climate change are realized in fisheries management systems. Changes in species distribution and trophic interactions, among other social and economic factors, will exacerbate the need for ecosystem considerations. The research by Kroetz will help prepare policymakers and managers for these changes and improve the ability to respond in an adaptive, proactive manner.

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4 Is Cap-and-Trade Causing More Greenhouse Gas Emissions in Disadvantaged Communities?

Kyle C. Meng (U.C. Santa Barbara and NBER)[1]

There is mounting concern that market-based climate change policies may be causing disproportionately more greenhouse gas emissions near disadvantaged communities. This paper uses facility-level reported greenhouse gas emissions data to examine the environmental justice consequences of California's cap-and-trade policy, the second largest such program in the world. I do not find that this policy has led to relatively more greenhouse gas emissions in disadvantaged communities. If anything, statistically noisy evidence suggests that emissions have fallen more in disadvantaged communities since the start of the program to 2015.

Introduction

Socially and economically disadvantaged individuals tend to experience more harm from environmental conditions (Mohai, Pellow, and Roberts 2009). Climate change is no exception. Recent studies on climate change impacts project considerable social and economic inequality under anthropogenic climate change, both across (Dell, Jones, and Olken 2012, Burke, Hsiang, and Miguel 2015, Dingel, Meng, and Hsiang 2018) and within (Burgess et al. 2013, Houser et al. 2015) countries. However, much less is known about how policies that mitigate climate change may themselves affect inequality. Because greenhouse gases (GHG) spread evenly around the planet and do not directly yield localized effects, the primary environmental justice concern is not about local GHG emissions per se. Rather, GHG emissions are often co-produced alongside local pollutants, such as particulates, carbon monoxide, and nitrogen oxides. Climate policies may lead to greater environmental inequality if they induce more GHG emissions near disadvantaged communities (Solomon and Lee 2000, Kaswan 2008, Stavins 2008, Ringquist 2011, Boyce and Pastor 2013).

In light of this, environmental justice concerns are now at the center of many climate policy debates. Recent efforts to introduce a carbon tax in the state of Washington failed in part due to environmental justice critiques.[2]

Similar debates are occurring about the European Union Emissions Trade System, the continent's flagship climate policy.[3] Nowhere has this issue been more hotly contested than in California, where the future of the state's pioneering cap-and-trade program has recently been questioned on equity grounds.[4] In particular, critics argued that the flexibility of using permit markets to meet regulatory requirements may allow polluting facilities near disadvantaged communities to increase GHG emissions.

Can cap and trade cause more GHG emissions in disadvantaged communities? Conceptually, there is no obvious answer. When functioning correctly, market-based incentives employed by cap and trade directs greater emissions reduction from cheaper, dirtier polluting facilities in the state. If such facilities tend to be located near disadvantaged communities, then these communities should see a larger decline in emissions under cap and trade than under non-market-based climate policies. Indeed, existing studies of another California cap-and-trade program for nitrogen oxide pollution have found that lower-income households are either not affected (Fowlie, Holland, and Mansur 2012) or may actually benefit from emissions trading (Grainger and Ruangmas 2018). On the other hand, if lower-cost facilities are not located near disadvantaged communities, then cap and trade may increase emissions from these facilities compared to a non-market-based regulation.[5]

This paper examines whether California's cap-and-trade program has caused relatively more GHG emissions in disadvantaged communities compared to other communities. To do this, I gather facility-level on-site GHG emissions data for all cap-and-trade regulated facilities in California since the start of the cap-and-trade program. I then compare average emission trends in zip codes that contain "disadvantaged communities," as defined by the California Environmental Protection Agency, against zip codes that do not contain such communities.

I do not find statistically significance evidence that California's cap-and-trade program has led to more GHG emissions in disadvantaged communities. The statistically noisy difference in average emission trends suggests that disadvantaged communities have seen a greater decline in GHG emissions since the start of the program in 2013.

Data Sources

Emissions

I obtain facility-level GHG emissions data collected under California's Regulation for the Mandatory Reporting of Greenhouse Gas Emissions from the Air Resources Board.[6] This data covers the 2012-2015 period.[7] To calculate a greenhouse-gas-emissions variable that corresponds most closely to emissions produced at the facility, I take the sum of GHG emissions (from biogenic and non-biogenic sources) resulting from on-site combustion.[8] Emissions data is then aggregated to the zip code level.

Zip code definition of a "disadvantaged community"

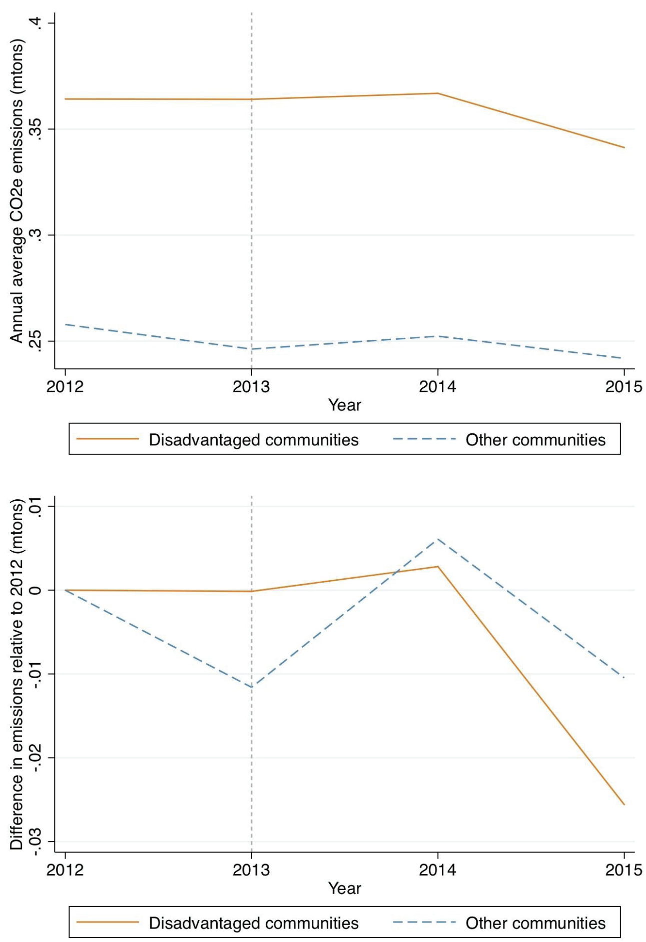
I use the legal definition of a "disadvantaged community" following California Senate Bill 535.[9] Specifically, I follow the California Environmental Protection Agency and Air Resources Board definition of a zip code as being disadvantaged if it contains all or part of a "Disadvantaged Community Census Tracts" with a CalEnviroScreen score in the top 25th percentile.[10]

Results

The top panel of Figure 1 shows average emissions for disadvantaged and non-disadvantaged zip codes. Throughout 2012-15, average on-site emissions (in annual megatons of CO_2 equivalent) produced in zip codes with disadvantaged communities were consistently higher than average emissions in zip codes that do not contain disadvantaged communities. This is consistent with a prior study showing that disadvantaged communities tend to be located near a greater number of GHG-emitting facilities and near the largest GHG-emitting facilities (Cushing et al. 2018). However, such evidence does not necessarily imply that cap and trade has changed environmental inequality.

The evidence in the top panel of Figure 1, however, does not support the argument that cap and trade has led to relatively more GHG emissions near disadvantaged communities. That key question is fundamentally about whether cap and trade has caused the gap in nearby emissions between disadvantaged and other communities to converge. To test that claim, I calculate the change in average emissions for each group relative to 2012 levels, the year just before the start of the cap-and-trade program, and examine whether disadvantaged and non-disadvantaged communities have experienced different emission trends. A greater drop in emissions for disadvantaged communities compared to other communities would suggest that cap and trade has caused emission differences to narrow across the two groups.

Figure 1. Emission trends for disadvantaged and other communities (2012-2015)



Notes: Top panel shows average annual on-site GHG emissions produced in zip codes that contain (orange, solid lines) and do not contain (blue, dashed line) a disadvantage community over 2012-2015. Bottom panel shows change in average emissions since 2012.

These trends are shown in the bottom panel of Figure 1. By and large, the annual change in emissions across disadvantaged and non-disadvantaged communities look similar. Over the 2012-15 period, it appears that emissions in disadvantaged communities have declined slightly more than that of other communities. However, a t-test does not detect a statistically significant difference in average 2012-15 GHG-emission trends across disadvantaged and non-disadvantaged communities. That test, conducted for subsamples with unequal variances, shows a mean difference of -0.013 megatons of GHG emissions per year with a t-value of 0.53.

Conclusion

Pollution tends to hurt disadvantaged communities the most. As such, existing environmental policies should be carefully evaluated on environmental justice grounds. Using recent facility-level emissions data, I do not find statistical evidence that California's cap-and-trade program has led to more greenhouse gas emissions in disadvantaged communities during 2012-15. If anything, the evidence suggests that disadvantaged communities may have experienced on average a greater decline in emissions since the start of the cap-and-trade program than other communities. This finding, however, does not obviate the need for additional policies that more directly address environmental justice concerns associated with local pollution. Such policies should exist in tandem with California's existing cap-and-trade policy.

Endnotes

1 I thank Irena Asmundson, Severin Borenstein, Jim Bushnell, Chris Costello, and Meredith Fowlie for helpful comments. All errors are my own.
2 Leber, Rebecca. October 31, 2016. "The Most Dramatic Climate Fight in the Election is in the State of Washington." *Grist*. http://grist.org/election-2016/washington-carbon-tax-732/.
3 "It is Time to Scrap the ETS!" Transnational Institute. https://www.tni.org/files/download/scrap_the_ets18feb.pdf.
4 Megerian, Chris. March 13, 2017. "In the Battle Over California Climate Policies, Green Projects Are Now In the Hot Seat." *Los Angeles Times*. http://www.latimes.com/politics/la-pol-ca-offsets-environmental-justice-20170313-story.html.
5 Additionally, it is possible that the distribution of more free permits to polluters near disadvantaged communities increases pollution over a longer period by subsidizing those polluters to stay in business.
6 Mandatory GHG Reporting - Reported Emissions. California Air Resources Board. Available here: https://www.arb.ca.gov/cc/reporting/ghg-rep/reported-data/ghg-reports.htm.
7 Mandatory Reporting of Greenhouse Gas Emissions data is available since 2008. However, 2008-10 and 2011-15 data are not consistently reported and thus cannot be directed compared. Furthermore, in 2011 there were potential issues with potential double-counting of emissions from natural gas distribution. As a consequence, I restrict our sample period to 2012-15.
8 This is also known as "Scope 1" emissions. I exclude "Scope 2" emissions associated with purchased electricity and emissions contained in the fuel sold by fuel suppliers.
9 This is the definition used for distributing cap-and-trade auction revenue funds to disadvantaged communities.
10 Maps to Support the Disadvantaged Communities Investment Guidelines. California Air Resources Board. Available here: https://www.arb.ca.gov/cc/capandtrade/auctionproceeds/535investments.htm

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RESPONSE

Larry Goulder (Stanford University)

The environmental impact of California's cap-and-trade program for disadvantaged communities has been an issue of considerable interest and controversy. The environmental justice community has expressed great concern that the cap-and-trade program implemented in 2012 as part of California's key climate policy Assembly Bill 32 has had an adverse impact on these communities.

Kyle Meng concentrates on the potential implications for greenhouse gas (GHG) emissions. A t-test derived from data over the period 2012 through 2015 finds no statistically significant evidence that GHG emissions have increased in disadvantaged communities relative to other California communities. The author might have performed a more extensive difference-in-differences approach, employing data prior to as well as subsequent to 2012. However, the simpler assessment in the paper suggests that a more elaborate assessment also would not yield statistically significant evidence of a larger impact in the disadvantaged communities.

The paper does not use theory to consider under what conditions the impact of GHG emissions on disadvantaged communities would be larger. A relevant condition would be that the GHG-emitting facilities in disadvantaged communities involve marginal abatement costs that rise more slowly than the marginal abatement costs of such facilities in other communities.

Of course, the main focus of the environmental justice community is not on the GHG emissions impact but rather on the emissions of local air pollutants (in particular, sulfur dioxide, nitrogen oxides, and particulate matter) associated with GHG emissions. The present paper is a prelude to a new paper by Meng and Danae Hernandez-Cries that considers this important issue. The authors intend to apply a difference-in-differences approach to evaluate this issue. A focus on the production technologies could offer a complementary empirical approach. If the production processes in the disadvantaged communities tend to involve stronger substitutability of local pollutants for carbon dioxide than the substitutability inherent in the production processes elsewhere, cap and trade would produce larger local-pollution impacts in the disadvantaged communities, other things equal.

It is useful to keep in mind that the issue of relative local-pollution impact can be subdivided into two issues, each of which is relevant to environmental justice. One is whether cap and trade has widened the gap between the disadvantaged communities and other communities, compared with the gap that would have occurred if cap and trade had not been implemented and business-as-usual conditions had continued. The other is whether cap and trade has failed to reduce the local pollution gap as much as some other policy might have. The environmental justice community often focuses on this latter issue, claiming that more direct controls on local pollutants are essential. California addressed environmental justice concerns by accompanying Assembly Bill 398, which extended cap and trade to 2030, with Assembly Bill 617, which promises direct regulation to address local pollution in disadvantaged communities. Without Assembly Bill 617, Assembly Bill 398 would not have passed, and the future of cap and trade in California would have remained in doubt.

5 Distributional Effects of Conservation Easements

Dominic Parker (University of Wisconsin and PERC)

Governments have long acted to protect land from development, sometimes through direct acquisition and other times through land-use regulation. But less centralized, incentive-based approaches are becoming more common across the globe. The U.S. system of preferential tax treatment toward conservation easements held by private land trusts is a leading example of decentralized conservation. In it, the government's main role is to set tax policy and then let private individuals and organizations, under limited regulation, determine the quantity and patterns of permanent conservation.

Easements are a private and voluntary form of land-use zoning. They are legally binding agreements through which landowners give up rights to subdivide and develop rural land but retain rights to farm.[1] The amount of U.S. acreage under easements has expanded from about 1 million acres in 1990 to more than 15 million acres today. This growth far outpaces that of more traditional approaches, such as lands set aside in national and state parks (see Parker and Thurman 2011, 2018).

The growth of easements is enabled by broad support from both the political right and left.[2] Most conservatives prefer land trusts to traditional government-led approaches, and liberals tend to view land trusts as an effective complement to the more traditional government approaches. Because of this broad support, in recent years, federal and state politicians have extended or added tax benefits to those who donate easements, thereby propelling growth.

Tax Policy under Scrutiny

As the use of easements has grown, however, the broad political support may be eroding. Critics have begun to raise questions about the distributional effects of conservation easements as well as their on-the-ground effectiveness. Although millions of acres are under easements, surprisingly little is known about their long-run impacts on land use in aggregate. Do easements actually provide additional conservation? Or do they simply displace development and reward wealthy landowners for actions they would take regardless of whether the land is under easement? What benefits do easements convey to ordinary Americans, many of whom are surprised to learn that most easements do not allow for public access on conserved lands?

These questions are relevant to public policy because the public is invested in the success of conservation easements. Although many easements are donated to land trusts, the donation is in exchange for tax breaks that U.S. taxpayers implicitly fund through foregone claims to tax revenue. Yet evaluating the public's return on easement investments is difficult because some of the activity is not transparent. For example, land trusts have not been required to report (or learn) the tax deductions claimed by donors, so information on the cost of easements is not readily accessible (see Looney 2017).[3]

The public does know the ranks of easement donors includes wealthy Americans. According to *Wall Street Journal* reports, some of the highest-valued easements cover golf courses owned by Donald Trump.[4] Large easements have also been donated by moguls such as Ted Turner and Tom Brokaw, to name a few.

The average easement donation was $475,416 over 2003-12, which dwarfs in value every other form of charitable giving on a per-donation basis, including art, real estate, and money.[5] IRS summary data show that while 2 percent of easements came from taxpayers with annual incomes greater than $10 million, these donors accounted for 23 percent of the monetary value. Taxpayers with incomes exceeding $500,000—roughly the top 1 percent—accounted from 17 percent of donation quantity but 75 percent of monetary value. Roughly 63 percent of easement donations came from taxpayers earning less than $200,000 in 2012. These statistics indicate that wealthy donors dominate in terms of deductible value, but also that the majority of donors have relatively modest incomes.

Donation Prices

Do wealthy donors dominate because they are more generous, or because they pay a lower donation "price"? To shed light on this question, I have developed a "conservation tax calculator" with collaborator Walter Thurman (see Parker and Thurman 2018). Conditional on a taxpayer's income, state of residence, and the year and value of a donation, the calculator estimates an after-tax donation price. This price incorporates federal and state income tax rates, rules about charitable deductions, and state tax credit programs. It also accounts for the dynamic effects of carryover provisions and annual income limits on easement deductions.

Figure 1 shows the price over 1987-2012 for an easement donated in the seven states without an income tax. Focusing on these states isolates and highlights the role of federal income tax policy. Each panel plots the price for four different taxpayers: those with adjusted gross incomes of $100,000, $200,000, $350,000, and $1 million. Panels A and B show calculator output for easements appraised at $500,000 and $1 million, respectively.

The prices demonstrate the combined effects of differences in marginal tax rates and of AGI limitations on deductions and carry-forward limits. Prior to 2006, the price increased with donation size primarily because of a five-year carry-forward limit on charitable deductions. Because of the AGI limits and the carry-forward constraints, a taxpayer with an AGI of $100,000 could deduct only $30,000 (0.30 x $100,000) each year for six years, leading to a total deduction of $180,000. Moreover, deductions in later years yield declining financial benefits due to the 5 percent annual discount rate we apply to the calculations. The price falls for the lower income donors in 2006 because the carry-forward period was extended from five to 15 years. The AGI limitation was also increased in 2006 from 30 to 50 percent for all donors and from 30 to 100 percent for qualifying farms and forests. Hence, a qualifying landowner with an AGI of $100,000 would fully exploit the $500,000 donation in 5 years, which lowers the price of conservation from 0.94 to 0.89 cents in terms of the after-tax cost of each dollar of value donated.

The main takeaway is that there is a gap in the donation price that increases with taxpayer income, holding donation size constant. Individuals with higher adjusted gross incomes effectively pay a lower after-tax "price" to donate a conservation easement. There is a gap in part because high-income donors pay higher marginal tax rates, and in part because land-rich but cash-poor donors have not been able to deduct the full value of their donation, especially prior to 2006. The gap grows further if we account for savings from state income taxation, which can be large in states with high marginal tax rates (e.g., California) and in states with generous tax credits for easement donors (e.g., Colorado).[6] The gap also grows further when we consider potential tax savings from income earned on capital gains, property taxes, and inheritance taxes. The gap grows further still if assume that wealthy landowners have better access to savvy tax advisors and real estate appraisers. (Savvy appraisers may be able to inflate easement donation values without IRS penalty if, as suggested in a recent Fortune magazine article highlighting tax abuse, the IRS is too resource-starved to audit returns.)[7]

For better or worse, market forces are starting to channel easement tax breaks to taxpayers who can fully exploit—and perhaps "over-exploit"—the tax incentives. Consider the recent rise of so-called syndicated easements. The deals involve groups of investors forming partnerships that buy land with the intent of donating easements for financial gain. These arrangements enable full tax-benefit exploitation because, with a large number of partners, each partner can claim the tax benefit without hitting the 50 percent of AGI tax constraint. Hence, full tax sheltering is one economic rationale for the emergence of syndicates. Another rationale is that syndicated easements are attractive to investors primarily because the deal brokers have access to appraisers who are willing to exaggerate donated values; this is the issue of focus in Looney (2017) and described in media outlets (see, e.g., Elkind 2017).

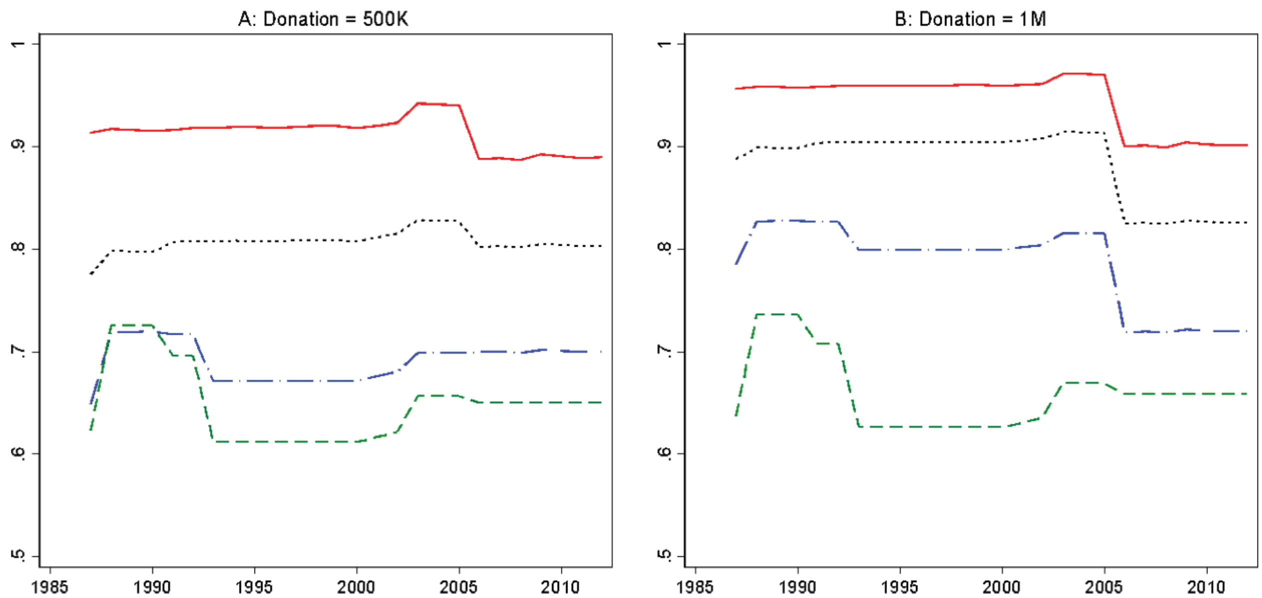
Dead Hand Control

Through their support of easements, federal and state tax codes encourage "dead hand control" of land because they require restrictions to be permanent and, unlike other forms of donation, are not subject to reversal (Mahoney 2002, McLaughlin 2005). This requirement is lauded by many environmentalists and land trusts but it is inconsistent with centuries of common law, which tends to discourage perpetual constraints on land use. One reason is that restrictions that freeze land use to a landowner's present desires may become antiquated and inefficient over time.[8] Freezing future land uses to the desires of wealthy land barons also raises distributional concerns. Such restrictions could exacerbate inequality in land ownership over time by preventing subdivision into parcels that would be affordable for middle-class Americans.

A More Equitable (and Effective) Public Financing Mechanism?

Elsewhere I have argued that relying on tax incentives may encourage *ad hoc* patterns of conservation that are expensive and ineffective in aggregate (Parker 2005). This is because the tax-code funding vehicle puts donating landowners in the driver's seat and weakens the ability (and incentives) of land trusts to use taxpayer dollars in ways that best serve the public's interest. Specifically, I have asked if land trusts could successfully self govern, for example, through Land Trust Alliance (LTA) accreditation, or if the movement would benefit from top-down federal regulation.

Figure 1. After Tax Price of Donating a Conservation Easement (one minus the proportion of easement value recovered through federal income tax savings)



Notes: The legend is as follows. AGI $100,000 is the red solid line. AGI $200,000 is the black dotted line. AGI $350,000 is the blue long dash-dotted line. AGI $1,000,000 is the green dashed line. We assume the AGI $100,000 and AGI $200,000 donors are qualified farmers and the higher AGI donors are not. All scenarios assume that none of the donation's value would otherwise be subjected to capital gains taxation. (If it was subjected to capital gains taxation, the after-tax price would be lower).

The continued growth of easements in general, and the rise of syndicated easements in particular, has put these same questions back in the spotlight. What kinds of reforms are needed? Can Land Trust Alliance accreditation alone weed out rogue land trusts that seek only to maximize tax breaks? Should the IRS cap the size of easement deductions?

Such a cap on the size of donations is proposed in the Charitable Conservation Easement Program Integrity Act of 2018. From my perspective as an economist, this proposed policy and others focused on IRS oversight would not remedy the fundamental incentive problem with donated easements. Because land trusts accepting donations do not use their own money, they are not fully incentivized to carefully compare the costs and benefits of conserving a proposed donation to alternative acquisitions. This is a reason why donated easements tend to be of inferior quality when compared to land that is purchased by land trusts (see Parker and Thurman 2018).

In Parker (2005), I contemplated an alternative public funding mechanism that could fix the incentive problem. Federal tax code funding for easements could be replaced by an equivalent level of funding through federal competitive grants requiring trusts to raise matching funds from private sources and local governments. This may seem like a radical step, but the potential benefits to the general taxpayer, who is already paying for easements, are worth considering.

First, the alternative scheme would allocate federal dollars to areas about which there is some consensus that the value of conservation is particularly high. Supporters of the trust would pay the most and therefore guide the decision; proposals could also be evaluated by land conservation experts. This process is in contrast to the current situation where a local landowner (including partnership syndicates) who has an interest in donating an easement triggers the process.

Second, once trusts received grant monies, they would have a budget constraint that would encourage them to act as if they bore the full costs of acquiring different parcels. Currently, the source of forgone tax revenue is virtually limitless, as long as the trust can find willing landowners. In contrast, fixed allocations of grant dollars would encourage trusts to prioritize.

A third and related advantage is that land trusts would have stronger incentives to oversee the appraisal of easements. Land trusts receiving federal grants would be using money from their own budget to buy easements, so they would be motivated to question appraisals that seem unreasonably high. More generally, instead of seeking easement donors from a small pool of parties motivated by tax incentives, land trusts would have a public funding source that allowed them to negotiate with a larger pool of potential easement sellers.

Looney (2017) proposes a reform that would convey similar advantages. His idea is to "take the deduction and transform it into a credit allocated to a donee organization. In this model, donee organizations would be empowered to approach landowners to 'spend' the credit and to decide what kind of properties to conserve and how much to pay. Because these organizations have the right incentives to conserve properties with the greatest environmental or historic value, this approach is intended to maximize the return on the tax benefits provided without requiring adversarial IRS oversight."

The key to both alternative funding vehicles—competitive grants or allocable credits—is that each would empower land trusts with more discretion (and funding), thereby putting them in the driver's seat. Ideally, land trusts would then prioritize lands for acquisition based on inherent conservation value and acquisition cost, independent of the owner's wealth or motives. Unlike the current system under tax code funding, land trusts would not need to prioritize their efforts based the degree to which the land's owner is incentivized to donate, which depends on landowner income and creativity in exploiting tax advantages. To be sure, either funding reform would introduce new challenges and problems not considered here. The benefit, however, is that each reform could delink conservation from tax considerations that may not align with cost-effective conservation.

Endnotes

1 Easements enable conservationists to buy and retire other rights from landowners—for example, the right to mine and log—and can be customized on a case-by-case basis.

2 The website of the Land Trust Alliance highlighted this, arguing that "bipartisan support for conservation easements exists because politicians know that this program works and brings important benefits to communities throughout the country."

3 According to Looney (2017), some land trusts voluntarily disclose the value of the donations they receive, but most do not.

4 See, for example, Rubin, R. (2017) "When a Conservation Tax Break Protects Backyards and Golf Courses." *Wall Street Journal*, June 1. Available at: https://blogs.wsj.com/economics/2017/06/01/when-a-conservation-tax-break-protects-backyards-and-golf-courses/

5 During the 2000s, the average value of a donation was $163,000 for land, $45,000 for stocks and other financial gifts, $37,000 for intellectual property, and $7,000 for art (Eagle 2011).

6 The tax credits in some states, such as Colorado and New Mexico, are tradable. Making credits tradable shrinks the gap in donation price between high and low income donors because cash-poor but land-rich owners can sell tax credits to higher-income taxpayers who can fully exploit the tax benefits.

7 Quoting the article: "The IRS has policing power, and it wields that clout chiefly by auditing the returns of those who take the deductions. But that's a torturously slow process and one that so far has yielded minimal results. The speed at which [alleged tax abuses of easements] have increased has left the resource-starved agency looking like a befuddled mall cop lurching off his chair and trying to figure out which of the dozen teenagers simultaneously grabbing candy bars to

chase down." See Elkind, P. (2017) "The Billion-Dollar Loophole." Fortune Magazine, Dec. 20. Available at: http://fortune.com/2017/12/20/conservation-easement-tax-deduction-loophole/.

8 Judged on the narrower criterion of effective conservation (rather than social efficiency), it is also not obvious that perpetual constraints on land use is good policy. As economic and ecological conditions change, the benefits and costs of conserving different parcels will change. It is doubtful that every easement currently held by land trusts will continue to yield conservation benefits in the face of population growth and migration, changing demands on agricultural land, climate change, and changes in preferences toward the preservation of different wildlife species.

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RESPONSE

Mike Conner (Director of Land Protection, The Nature Conservancy)

Nick Parker discusses several things that I think warrant a discussion, such as issues with easements being perpetual and assumptions around appraisals and who leads the conservation effort. I will focus here on the primary problem and solution recognized in the paper.

The problem identified is opportunistic acquisition: that tax benefits for the wealthy drive easement donations rather than conservation concerns, with the result being less strategic acquisition or opportunistic acquisition.

The solution proposed is to grant tax deductions to landowners through competitive grants to land trusts; reasoning that by combining such grants with additional investment on the part of land trusts, the land trusts will be encouraged to decide which parcels to invest in, resulting in the protection of properties with higher conservation values.

I like the idea of creating a formula that will result in more disciplined acquisition for conservation.

While opportunistic acquisition may have once been used extensively by the land trust community, and still may be used by very small land trusts, in more recent years conservation easement acquisition by many large land trusts has shifted to more of a purchase model whereby easements are purchased by land trusts. Many landowners have recognized that a sale of a conservation easement is more lucrative to them than the write-off.

Under the purchase model, the land trusts negotiate the terms of the conservation easement, get an appraisal, pen the terms of the deal, and then raise the funds from public and private sources to buy the easement. On top of that, land trusts that want accreditation from the Land Trust Alliance now have the obligation to raise funds for a stewardship endowment. At a minimum, the non-wasting endowment must pay for all costs related to annual compliance monitoring.

Similar to Parker's suggested solution, this investment of time and financial resources forces land trusts to prioritize acquisitions according to their missions. Furthermore, often there are layers of priorities—when the Nature Conservancy applies for state funds, for instance, our project must advance the priorities of the state program to receive the grant. The same is true for many foundations.

The federal tax deduction is still important because one might be able to negotiate a bargain sale or a discounted purchase price where the landowner writes off the difference between the appraised price and the sale price. But I would argue that the investment of the land trust in both the purchase price and the stewardship endowment forces acquisition discipline.

It would be extremely useful to figure out how to use the federal tax benefit to help offset stewardship expenses to protect an easement-encumbered area. Better enforcement by the IRS is needed to ensure that appraisals accurately represent the value of the donation and write off and that the conservation benefits are genuine.

6 Transferable Conservation Obligations in Patchy Systems

Andrew Plantinga (U.C. Santa Barbara) and Christopher Costello (U.C. Santa Barbara, NBER, and PERC)

Introduction

We study the institutions governing decentralized natural resource use in a spatial, patchy environment. A patchy system is characterized by homogeneity within spatial units ("patches") but heterogeneity across units. If the resource moves across patches, as with water, migratory fish and wildlife, and invasive plant species, there would appear to be formidable challenges to efficient management of the resource. In general, the extraction of a unit of the resource in one patch changes conditions in all other patches. These spatial externalities can include effects on extraction costs and production of the resource. Furthermore, the complexity of the management problem may be compounded by economic heterogeneity (different costs and benefits among patches) and biological heterogeneity (different productivity among patches). As has been shown by Kaffine and Costello (2011), efficient management requires incentivizing resource users to take particular actions in each patch and to coordinate these actions among patches to achieve system-level efficiency.

The literature on tradable ambient pollution permits may provide helpful guidance. In the standard setting, there are J emissions sources and M receptors, and the effect of a unit of emissions at source j on ambient pollution at receptor m is given by a transfer coefficient a_{mj}. Transfer coefficients are analogous to inter-patch water flows and dispersal rates for animal and plant species. As shown by Montgomery (1972), optimal pollution control can be achieved with a system of tradable ambient pollution permits. The regulator issues a set of L_m permits, one for each of the M receptors. Permit trading gives rise to an equilibrium price for receptor m permits equal to the marginal damages from pollution at receptor m. Although tradable permits can generate an efficient solution for a patchy system, the mechanism is still quite complex. Separate permits are needed for each receptor, and the permits need to be traded in separate markets. As well, trading ratios need to be incorporated in each of the permit markets if marginal damages of emissions differ among sources.

We consider a mobile renewable resource that is extracted by users holding within-patch property rights to the resource. Thus, users choose the optimal within-patch harvest but ignore the effects of their extraction on other patches. There is heterogeneity among patches in productivity and extraction costs. We introduce a system of tradable conservation obligations (TCOs), which require that a minimum level of environmental quality (or "conservation") be maintained at any given time. Examples include obligations to leave fish in the ocean (escapement) or water in a river (instream flow). We show that when the resource in each patch disperses to at most one other patch, efficiency can be achieved through a series of bilateral negotiations. In these cases, the incentives are in place to allocate conservation efficiently among patches, and the role for TCOs would potentially be to reduce transactions costs and address distributional goals. In contrast to the model in Montgomery (1972), resource users reap the benefits of conservation in other patches, which gives rise to incentives for Coasean bargaining. However, under more general dispersal conditions, we show that conservation is a public good and that free-riding would undermine the incentives for efficient bargaining. We conjecture that, in these cases, TCOs can produce a better second-best outcome than would be achieved with decentralized bargaining.

The Model

A spatially connected system is exhaustively composed of a set of I discrete patches, with index $i=1,...,I$. The renewable resource stock residing in patch i at the beginning of period t is given by x_{it}, harvest during period t is h_{it}, and this leaves residual stock e_{it}. Price is constant (p), but marginal harvesting cost is given by $c_i(s)$, so the marginal harvest cost depends on the contemporaneous resource stock. If the starting stock is x and the residual stock is e, then total revenue over period t is $p(x\text{-}e)$, and total cost is the integral of $c_i(s)$ from e up to x. Using a trick first noticed by Reed (1979), let $\bar{e}_i$ denote the solution to $p = c(\bar{e}_i)$. This is the residual stock level that returns exactly zero marginal profit. Define $Q_i(S) \equiv \int_{\bar{e}_i}^{S}(p - c_i)(s))ds$, so $Q_i(S)$ is the annual profit from harvesting a stock S all the way down to the point where it is no longer profitable to extract any more. Then profit during period t can simply be written

$$Profit_{it} = Q_i(x_{it}) - Q_i(e_{it})$$

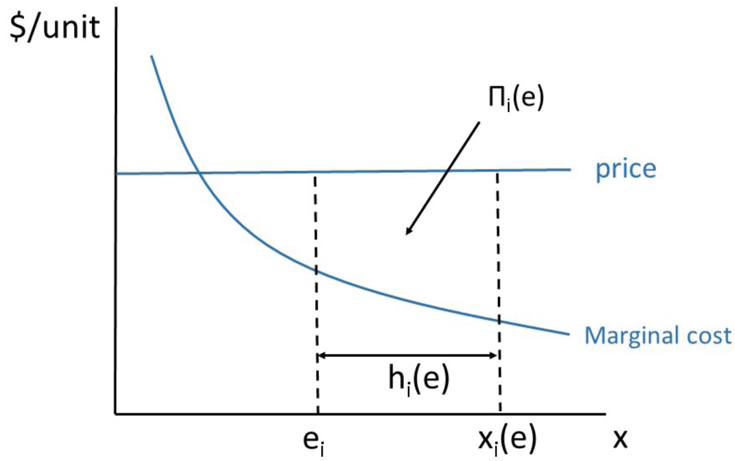
Although not considered here, the resource stock could potentially generate an *existence value*, given by $Z_i(e_{it})$, so the total economic value of patch i, period t, would be $Q_i(x_{it}) - Q_i(e_{it}) + Z_i(e_{it})$.

Biological reproduction in patch i depends on the residual stock. The stock produced in patch i is $g_i(e_i)$, which then disperses around to other patches. Dispersal is described by the I x I matrix D with elements D_{ij} indicating the share of the residual stock that moves from patch i to j. The equation of motion that defines the resource stock in patch i at the beginning of period $t+1$ is:

$$x_{it+1} = \sum_{j=1}^{I} g_j\,(e_{jt})D_{ji}$$

In steady-state, $x_{t+1}=x_t$ and so steady-state profit is a function solely of steady-state residual stocks $e=(e_1,e_2,...,e_I)$. We denote steady-state profits from patch i as $\pi_i\,(e)$ (see Figure 1 for an illustration). The TCO will be defined over units of residual stock. Formally, the TCO is a requirement to conserve a specified amount of the resource $\tilde{e} = (\tilde{e}_1,...,\tilde{e}_I)$.

Figure 1: Steady-state stock, harvest, and profits in a single patch.



We focus here on steady-state solutions in which optimal management of the resource maximizes the sum of profits from all of the patches:

$$\pi_i(e) = \sum_{j=1}^{I} \pi_i(e)$$

The first-order conditions are:

$$\sum_{j=1}^{I} \pi_{ji}(e) = 0 \text{ for all } i$$

where $\pi_{ji}(e)$ is the partial derivative of profits in the jth patch with respect to residual stock in the ith patch. We identify the conditions under which decentralizing bargaining or a system of TCOs can reproduce the first-order conditions.

Results

To set ideas, consider the two-patch case ($I = 2$). We assume that the two-patch owners maximize profits in their own patches but ignore the benefits that spill over to the other patch. Thus, steady-state residual stocks satisfy $\pi_{11} = 0$ and $\pi_{22} = 0$. For both users, maintaining a larger residual stock in their own patch is costly because harvest must be reduced. However, the larger residual stock increases growth of the resource. Some of this additional growth remains in the patch, but some of it disperses to the other patch. The marginal cost to user 1 of increasing the residual stock in patch 1 is $-\pi_{11}$ and the marginal benefit to patch 2 is π_{21}. Similarly, the marginal cost to user 2 of increasing the residual stock in patch 2 is $-\pi_{22}$ and the marginal benefit to user 1 is π_{12}. In this simple case, all of the benefits and costs of conservation are internal to the two-patch users. As long as the costs of negotiation are not too high, the two users have an incentive to bargain to an efficient solution. In particular, user 2 would be willing to pay for more conservation in patch 1, and user 1 would be willing to provide it, up until $-\pi_{11} = \pi_{21}$. Similarly, bargaining over conservation in patch 2 will cease when $-\pi_{22} = \pi_{12}$. Thus, bargaining reproduces the first-order conditions for optimality in the case of $I = 2$.

When $I > 2$, the decentralized bargaining result holds with restrictions on the dispersal matrix D. Specifically, there can be at most one non-zero off-diagonal element of the matrix (non-zero diagonal elements are allowed). Examples include a source-sink model in which all patches flow to a single patch (for example, streams that all flow into a lake) and a circular (or linear) flow model in which each patch flows to at most one other patch (for example, fish movement with ocean currents). The important feature of these cases is that the residual stock left in a given patch benefits only one other patch, allowing for bilateral negotiations among patches as in the two-patch model described above. As an illustration, consider the three-patch circular flow model with non-zero off-diagonal elements D_{12}, D_{23}, and D_{31}. The conditions for optimality are:

$$-\pi_{11}(e_1,e_3) = \pi_{21}(e_1,e_2)$$
$$-\pi_{22}(e_1,e_2) = \pi_{32}(e_2,e_3)$$
$$-\pi_{33}(e_2,e_3) = \pi_{13}(e_1,e_3)$$

We conjecture that these conditions can be satisfied through a series of bilateral negotiations. Suppose that patch users 2 and 3 commit to e_2 and e_3. With these residual stock levels fixed, users 1 and 2 negotiate over e_1, arriving at the first condition. Now e_1 and e_3 are set and users 2 and 3 negotiate over e_2, arriving at the second condition. Under the right conditions on the profit functions, this process should converge to an equilibrium corresponding to the three optimality conditions, as in Cobweb and Cournot models.

When incentives are in place for efficient bargaining, TCOs may have a role to play in reducing transactions costs and addressing distributional issues. Consider a source-sink model where all patches $i > 1$ flow to patch 1 and I is large. In this case, the user of patch 1 would need to conduct $I - 1$ separate bilateral negotiations with the other patch users. Alternatively, TCOs would codify the conservation obligations for each patch user, facilitating the emergence of a market for TCOs. In equilibrium, TCOs would trade at a single price equal to π_{ij}, the marginal benefit to the sink patch from conservation in patch j. In addition, the initial distribution of TCOs determines which patch users are buyers and sellers of obligations. In contrast to a regulation requiring patch owners to increase conservation without compensation, TCOs can be allocated to achieve a more even distribution of the total gains from resource use.

When the residual stock benefits more than one patch, incentives for free-riding arise. To see this, consider the first-order condition for e_1 in the general case:

$$-\pi_{11} = \sum_{j=2}^{I} \pi_{j1}(e)$$

The benefits from the residual stock are non-excludable, which means that users in patches 2 through I benefit from residual stock left in patch 1 regardless of whether they pay for it. Under these conditions, users have the incentive to free-ride on the contributions of others, with the result that payments to the patch 1 user are unlikely to be sufficient to induce efficient conservation.

Can TCOs improve this outcome? Free-riding results in too little conservation, but the regulator of the TCOs system controls the total amount of conservation (the sum of the elements of $\tilde{e}$), which can be set at the efficient amount. Whether TCOs achieve the efficient allocation of conservation among patches is likely to depend on how heterogeneous the patches are. We conjecture that if patches are homogenous, the efficient solution will be obtained as long as the total amount of conservation (the "cap") is set at the efficient level. (Because TCOs involve obligations to do more conservation rather than less, a "floor" would be a more accurate term.) The idea is that if all patches are homogenous, then they will do the same amount of conservation, subject to the cap. This is the efficient solution when patches are homogenous. We conjecture that if patches are heterogeneous, TCOs can produce better second-best outcomes than with decentralized bargaining. With heterogeneity, TCOs still get the total conservation right, although not necessarily the right distribution of conservation among patches due to free-riding. There are likely to be cases where this outcome is closer to the optimum than what is achieved with decentralized bargaining.

Conclusions

We study a mobile renewable resource in a patchy environment. A main finding is that when the resource flows to at most one other patch, efficient use of the resource can be achieved through decentralized bargaining. The same result is not found with analogous cases involving tradable ambient pollution permits. For example, with a single pollution receptor and many sources, the solution in Montgomery (1972) is for a regulator to set a cap on total pollution and use trading ratios to ensure that marginal damages are equated across sources. In our setting, the incentives are in place for efficient bargaining because conservation provides benefits to resource users, in addition to imposing costs, which means that users have the incentive to bargain to where marginal costs and benefits are equal. In this case, TCOs can play a role in reducing transactions costs and achieving distributional goals. With more connectivity among patches, free-riding is a barrier to efficient bargaining. By getting the total amount of conservation right, it is likely that TCOs can achieve better second-best outcomes than would be obtained through decentralized negotiation. To study this issue more, we need a formal description of the free-riding equilibrium.

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RESPONSE

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Andrew J. Plantinga and Christopher Costello have described an interesting and important issue that has wide relevance for the management of renewable resources. They ask us to consider a situation where each person[1] can claim and manage a spatially delineated patch of a fishery.[2] He (or she) is the only one fishing in that patch. In any period his decisions to limit harvest allow the population to grow and, as a result, expand what is available in the next period. Fish swim, so some fraction of the next period's fish from his patch will migrate to other patches. This dispersal means those fishing in other patches will benefit from his conservation because *nature shares*. Of course, he is not alone. Fishers in other patches face the same situation. Some of the losses from their conservation are his gains. As Plantinga and Costello note, the steady-state profits in any patch are a function of overall the steady-state residual stock within this system as a whole. As a result, it will be based on the conservation decisions in all of the patches defining the fishery. So there is a clear coordination problem.

My comments will first discuss parallels between their problem and some aspects of models of charity as a public good as well as situations with reciprocal externalities. Then I will comment on some limitations in their model and possible areas for extensions.

Their discussion highlights opportunities for trading when there are only two agents or when the dispersal matrix limits the interactions to just two fishers. Suppose we consider their dispersal matrix a bit differently. Their equation (2) tells us that the stock in patch i is the result of *all* fishers' conservation decisions, growth in all patches, and resulting dispersal. Social goals to maximize the sum of all profits from all patches would take account of the *sharing*. Private actions of an individual fisher would not. They are efficient from his perspective but not from a social perspective. In Buchanan and Stubblebine (1962), they remain Pareto relevant.

If we highlight what fisher *i's* contribution to the stocks of others in comparison to each one of the others' contribution to him we may see some parallels to issues raised in the literature on charity as a public good. Using their notation, these terms are given in equations (1) and (2) below:

$$(1) \quad g_i\,(e_{it}) \cdot D_{ij}\,,\ j = 1,2,\ldots, I$$
$$(2) \quad g_j\,(e_{jt}) \cdot D_{ji},\ j = 1,2,\ldots, I$$

Notice the difference in the subscripts. In the first equation we are describing how *i's* conservation goes to j and in the second how *j's* conservation is shared with i. Plantinga and Costello begin their discussion of the distinction between private incentives and social efficiency with a two-patch system. With an external cap on the total residual stock, there are clear-cut incentives for trading.

There may be prospects for going a bit further by considering the analogy to matching in the pubic goods literature (Boadway et al. 1989, and Buchholz et al. 2012) in characterizing the properties of non-cooperative (non-degenerate) solutions in these cases. Matching rules can mitigate free-riding incentives and might be seen to reduce disparities between what i shares and j shares through externally imposed matching rates. Of course, each set of results is specialized. Nonetheless, this process would treat the sharing of others through dispersal to patch i as akin to flat contributions to his stock that could be supplemented at a fixed matching rate. Plantinga and Costello's case is more complex. Fisher *i's* increased conservation helps others (as indicated in equation (1) above for each *j*) and what they share with him (equation (2)). It is the "net" effect of nature's dispersal in equilibrium with different external match rates that is of interest.

More generally, Plantinga and Costello have offered an imaginative first step in what promises to be a rich new agenda for rethinking renewal resource issues. As they develop this new view they need to consider how their results are sensitive to some of the key assumptions. In many circumstances, fishers don't "stay put." They travel to others' patches, and the implied exclusive right one fisher has to manage his patch is compromised. Equally important, their proposed generalization to allow existence values for patches may need to be reconsidered for some applications. Existence (or non-use) values are not necessarily patch specific. One needs to consider what is unique about the patch or the stock that exists in a specific path. It may simply be the overall stock that has the existence value.

Thus, Plantinga and Costello's tradeable conservation obligations have clear promise as one policy instrument for assisting in managing some classes of renewable resources. When the policy alternatives expand to multiple goals—total conservation and the distribution among patches—we need another instrument, and some form of matching may fit the bill.

Endnotes

1 I will use the label of the individual here as a "fisher."

2 Of course it need not be a fishery or have a specific geography, but I will use the fishery case here and not deal with abstractions beyond a geographic area.

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7 Distributional Implications of Group Performance Mechanisms

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Introduction

Typically, market-based environmental instruments are levied at the level of the individual firm or pollution source. Examples include emissions taxes imposed on releases or discharges from a specific facility, pollution-based cap-and-trade programs where allowances are allocated or auctioned off to individual sources, individual tradable harvest quotas allocated to individual fishing vessels, and payments for ecosystem services (PES) made to individual landowners. However, market-based instruments can also be applied at the group or collective level. For example, rights can be allocated to a group (e.g., a group of polluters, vessels, or landowners) or rewards and penalties can be triggered based on group performance, rather than by individual performance. Examples include the allocation of a total allowable catch for a target species to a fishing cooperative, imposition of a fleet-wide limit on bycatch of a threatened or endangered species, use of an ambient tax or subsidy-based policy to control agricultural nonpoint source pollution, PES programs that pay communities rather than individual landowners for reduced deforestation, and voluntary environmental programs that are coupled with an industry-wide regulatory threat.

A key feature of group-level policies is that they create a policy-induced interdependence among the profits or payoffs to individuals within the group, which is distinct from—but possibly in addition to—any interdependence that might arise due to physical interactions (e.g., from congestion or joint production) or market interactions (e.g., through price). In other words, the cost or net payment for a group member depends not only on his or her own decisions but also on the decisions of others in the group. For example, under a collective harvest quota, use of part of the collective quota by one harvester implies that less is available for use by others, which implies that the opportunities available to one harvester are impacted by the decisions of others. This creates a policy-induced interdependence among group members and their net costs that has both distributional and incentive effects.

The rationale for using a collective approach varies across context but often relates to reduced monitoring or transaction costs or to the creation of incentives for information sharing, risk pooling, and within-group solutions to within-group externalities. However, group policies can also have disadvantages, including possible free-riding or shirking among group members, adverse selection into the group (if membership is voluntary), the impact of uncertainty about the behavior of other group members, and concerns about fairness (when one member of the group is penalized for the behavior of others). Thus, group policies raise a fundamental trade-off: They can promote efficiency and beneficial collaboration and coordination among group members, but at the same time, it can be difficult to sustain collaboration, and the benefits may be unevenly distributed across group members.

A growing literature seeks to understand the incentive effects of group-performance-based policy approaches (see, e.g., Kotchen and Segerson 2018, Zhou and Segerson 2016). This literature has focused on economic efficiency—for example, the design of policies to ensure efficient incentives for pollution reduction, fishing effort, or bycatch avoidance. This literature does recognize that policies designed to create efficient incentives may have distributional impacts that could be onerous for individuals and might limit their political acceptability. However, these distributional issues (and the associated "fairness" issues) have taken a back seat to the efficiency questions that have been the focus of most of the literature to date.

This essay provides an overview of some of the distributional issues related to group-performance-based market mechanisms. Although the principles discussed here apply across a wide variety of contexts, for concreteness, the latter part of the discussion will use collective approaches in fisheries to illustrate key points. An overarching message is that, while there are many parallels, the set of distributional (as well as efficiency) issues that arise in this context is broader than that associated with the application of market mechanisms at the individual level. Key reasons include the interdependence of payoffs created by group policies, possible incentives for collusion among group members, and the potential for multiple equilibria with differing distributional implications. Moreover, the distributional implications of group policies can in turn have efficiency implications, because they can affect the political feasibility of using a particular policy approach as well as the willingness of individuals to participate and/or cooperate in meeting environmental or resource management goals.

As a result, the distributional impacts of collective approaches depend not only on policy design but also on the way and extent to which the group cooperates internally. In other words, both the distributional and incentive effects of a group-performance-based policy hinge on a combination of the ecological, economic, and social features that affect group interdependencies within the group, the way in which the group organizes itself (including the internal "rules" it imposes on its members), and the design and nature of the collective rights that are granted (including any government-imposed restrictions and/or responses if the group meets or fails to meet specified objectives).

Distributional Issues under a Collective Approach

The key feature that distinguishes a group policy from one based on individual performance is that the "payment" hinges on group performance rather than individual performance or decisions. This broad class of policy approaches includes as special cases a wide range of market-based mechanisms, including a pure tax approach, a pure subsidy approach, a group-level allowable limit or quota (with some form of penalty for exceeding it), and various hybrid or combination approaches. Clearly, varying the specific policy parameters (such as the tax or subsidy rates or the performance standard) will change the total cost (i.e., net payments made or received) for each group member and hence the distributional impacts of the policy. Many of the distributional issues related to the choice of policy parameters (e.g., use of taxes vs. subsidies vs. freely allocated quotas) are analogous to those that arise under individual-based policies. However, the chosen policy parameters can also have distributional implications that would not arise when applied at the individual level.

For example, to avoid free-riding when group members make decentralized decisions, an efficient tax on group performance implies that each member of the group pays the full marginal damages from the group's environmental impact. This has several important distributional implications. First, total tax payments by each individual can be large, especially if the group is large. In addition, each group member is penalized for an increased environmental impact, regardless of who "caused" that increase, and each member's total tax payments will exceed its own incremental contribution to damages, thereby raising concerns not only about the total cost burden but also about fairness. These concerns can be reduced by allocating a "free" allowable limit or using a combined tax-subsidy approach instead of a pure tax approach.

Additional considerations can arise as well. For example, under some policy designs, members of the group have an incentive to collude to reduce tax payments or increase subsidy payments. The incentive to do so can be strong because the total tax or subsidy for each group member can be large. To avoid creating inefficient incentives to "game" the system, regulators need to anticipate these possible incentives and structure rewards and/or penalties accordingly.

One common group approach is the use of a collective limit or cap with a fixed penalty for exceedance. An example is a total allowable catch or fleet-wide bycatch limit in a fishery, where the fishery is closed when the limit is reached. A key feature of collective caps is that they can give rise to multiple equilibria (i.e., multiple ways of meeting the cap). This can create the potential for shirking or free-riding within the group and an unequal (and possibly unfair) distribution of the burden of meeting the cap. At the same time, it means there are potential gains for all member of the group from a more efficient allocation of the "effort" needed to meet the cap. However, whether these gains are realized will ultimately depend on the ability of the group to solve its own collective action problem.

Finally, when group performance is stochastically related to individual decisions, policy design can affect not only the distribution of costs but the distribution of risks as well. This can be an important consideration not only when individuals are risk averse but even when they are risk neutral. For example, if pooling quotas reduces the risk of exceeding quotas, all members of the group could potentially benefit even when they are risk neutral.

The above discussion suggests that group policies create a "local public good" that must be "managed" by the group in an effort to increase benefits or reduce costs (e.g., lower penalties, increase payments received, ensure regulatory relief). However, the potential gains from managing group performance efficiently are not necessarily evenly distributed. As a result, having gains from within-group collaboration is necessary but not sufficient for the success of a group approach. Both the distributional and efficiency implications of a group policy will depend on not only the policy design itself but also on the group's internal rules. The factors that are likely to contribute to success mirror the "design principles" identified by Ostrom (1990), which include group size and heterogeneity, leadership and social capital, and the group's ability to internally monitor and enforce its rules.

Collective Quotas in Fisheries

Although group policies have been used and/or suggested in a wide variety of contexts, one context where they are particularly widespread is rights-based management of fisheries (e.g., Zhou and Segerson 2016, Holland forthcoming). For example, Bonzon et al. (2013) report that 10 percent of catch share programs allocate allowable catch to a group, and a survey by Ovando et al. (2013) finds that 50 percent of surveyed fishing cooperatives faced a government-imposed collective cap. Examples of the use of collective quotas in fisheries include the Chignik salmon cooperative (Deacon et al. 2008), the New England groundfish sectors (e.g., Holland and Weirsma 2010), and the Bering Sea-Aleutian Island non-pollock groundfish trawl fishery (Abbott et al. 2015). These examples and others (see, e.g., Platteau and Seki 2001, Townsend et al. 2008, Holland forthcoming) illustrate the role that both the externally imposed policy (here, the collective limit) and the internal rules and operations of the group can play in determining the success of a group approach and its resulting distributional impacts. The following sections discuss a number of issues that arise in the fisheries context.

Species-based vs. Spatial Rights

In many cases, collective rights are tied to the harvest (or bycatch) of a specific species or collection of species, while in other cases, they are spatially delineated (Uchida et al. 2011, Wilen, Cancino and Uchida 2012). In the latter case, the rights granted within the designated space can be broadly defined (e.g., covering the entire marine system) or more narrowly defined to include only a subset of the marine resources within that space. These can have different distributional as well as incentive effects. For example, when species move across boundaries, spatially delineated rights can have spillover impacts that occur outside the designated area (Holland 2004, White and Costello 2011). Thus, decisions by one group can positively or negatively impact other groups.

Collective Rights/Limits vs. Collective Decisions

The extent to which the assignment of collective rights or responsibilities leads to collective *decisions* can vary significantly, ranging anywhere from full coordination (where, for example, a manager or committee makes all decisions for members of the group) to no coordination (where each member of the group continues to act independently despite the collective constraint). For example, the industry-wide sea turtle bycatch limit in the Pacific longline swordfish fishery imposes an aggregate limit on bycatch across all vessels, but vessel owners do not collectively manage the fishery or bycatch (Segerson 2010). Likewise, in many of the New England groundfish sectors, the sector-wide catch allocation is distributed to individual sector members, who then make independent decisions about how to use their share of the total quota (Holland and Weirsma 2010). In contrast, the short-lived Alaskan Chignik salmon cooperative was granted a collective allowable catch and managed that allocation cooperatively (Deacon et al. 2008). Similarly, the local fishing organization that manages the sakuraebi fishery in Japan makes most decisions cooperatively (Uchida and Baba 2008), as does the deep-sea crab fishery in New Zealand (Soboil and Craig 2008). The extent to which decisions by members of the group are coordinated can have important distributional implications (Zhou and Segerson 2016).

Types of Cooperative Activities

When members within a group choose to act cooperatively or collectively, they can do so in a number of ways (Uchida and Makino 2008, Deacon 2012, Ovando et al. 2013). These include limits on and coordination of effort and harvest activities (including gear restrictions), sharing of public inputs (e.g., gear, information, and infrastructure), cooperative marketing, and stewardship-related activities (including bycatch avoidance, conservation, and habitat enhancement). Which of these types of cooperation are likely to be most beneficial to the group will depend on the nature and source of the interdependencies within the group.

Income-sharing or Pooling Rules

In addition to engaging in cooperative behavior of different types, some groups also use a form of income sharing or pooling[1]. The potential benefits of pooling are usually described in terms of risk-sharing (or insurance) or the creation of incentives for cooperation leading to increased profitability (Platteau and Seki 2001). For example, pooling can reduce the incentive to "race to fish," while at the same time creating incentives to share information and some inputs. However, pooling can also generate a free-rider problem, under which members of a group face an incentive to shirk (Heintzelman et al. 2009). If members of the group bear the full costs of some activities but then reap only a share of the associated benefits, they will not face an efficient incentive to engage in those activities. Rather, they have an incentive to free ride on the efforts of others. Free-riding clearly has negative distributional implications for other members of the group.

A number of features relating to the design of the pooling arrangement affect its distributional (as well as efficiency) impacts. These include: (1) whether members only share revenue or also share some or all of their costs (and hence share profit); (2) whether sharing is full or partial, i.e., members pool all or only a fraction of their revenues and/or costs; and (3) how the proceeds in the pool are redistributed (for example, equally or according to some other pre-set criteria)[2]. For example, when members of the group share all revenues and costs, they have an incentive to maximize the total profit for the group (Deacon et al. 2008), but the ultimate distributional impacts will depend on how those profits are shared.

Finally, as noted above, a collective approach is sometimes advocated as a means of sharing or spreading risk[3]. For example, when harvesters cannot fully control the composition or quantity of catch or bycatch, pooling provides a mechanism for covering the excess catch by a member who "experiences a run of bad luck" (Deacon 2012). However, combining quota will not always lead to lower risk. Whether it does or not depends on both the nature of the underlying distribution of harvests and the magnitudes of the quotas. For example, under a uniform distribution, the probability of violating an individual limit is greater than the probability of collectively violating the combined quota when the quotas are relatively high, but the opposite is true when the quotas are relatively low (see Zhou and Segerson 2016).

Conclusion

Overall, this paper seeks to highlight the many distributional impacts and considerations that arise when a group performance-based policy is used to promote environmental or natural resource management. Those impacts are more nuanced and complex than the distributional impacts of individual-based policies. For example, the distributional impacts can be amplified when based on group performance. In addition, policies designed to address impacts outside the group can create incentives for collusion and risk sharing as well as multiple equilibria, all of which can lead to gains from collaboration. However, these gains can be unevenly distributed and suffer from the classic collective-action problem associated with managing any common property resource (here, group performance). The success and distributional impacts of a group approach will ultimately depend not only on the policy design (as determined by regulators) but also on the internal rules that govern behavior and decisions within the group.

Endnotes

1 Ovando, et al. (2013) report that 47% of surveyed fishing cooperatives engaged in some sort of proceeds sharing. For examples of cooperatively managed fisheries with different types of sharing rules, see, for example, Platteau and Seki (2001), Deacon et al. (2008), and Uchica and Baba (2008).

2 Revenue or profit sharing is often designed to ensure equal outcomes for all members of the group. Alternatively, collective management can be designed to provide equal opportunities, such as equal access to prime fishing locations or times. Some cooperative fisheries have used a rotation system to allocate opportunity equally across members of the group (see, for example, Uchida and Watanobe 2008). Rotations can be viewed as a sharing system based on sharing some inputs rather than sharing revenue or profits from outputs. However, unlike equal revenue or profit sharing, equally sharing access to inputs does not provide a means for risk sharing (Uchida and Baba 2008).

3 Harvesters subject to individual limits can also voluntarily combine quota to form risk pools as a means of managing the stochasticity of harvests. See, for example, Holland (2010), Holland and Wiersma (2010), and Holland and Jannot (2012). However, in a case study of three Japanese fishing groups, Platteau and Seki (2001) found evidence of pooling to manage risks related to inputs (nets) but no evidence that fishermen viewed pooling as an important means of smoothing income fluctuations.

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RESPONSE

Merrick Burden (Environmental Defense Fund)

Levying rights and responsibilities to a group typically means that policymakers are engaging in a form of co-management with that group. By granting rights and responsibilities at the group level, policymakers expect the group to undertake activities in ways that lead to improvements in social, economic, or conservation outcomes.

When a group receives a right and accompanying set of responsibilities, that group must take it upon itself to determine the sharing of benefits among group members. Likewise, the group must establish rules and responsibilities of individual members. Often, groups address the sharing of benefits by dividing group rights in such a way that individual members have individual rights, or similar. In this case, policymakers have not engaged in individual allocations, but the group—upon receiving an allocation from policymakers—determines the way in which to allocate at the individual level.

When policymakers allocate to a group, distributional issues among members are critical to determining the success of that group. Group members must collectively agree to the way in which the benefits of production (be it from fisheries, timber, or other industries) are shared. Striking an agreement is likely to be based on the ability of the group to determine a benefit-sharing arrangement that all group members find fair and equitable. Absent an allocation or sharing approach that group members agree to, the result of a group allocation may be simply a small version of a tragedy of the commons (e.g., a race for fish among members of a fishery sector or cooperative).

Certain group characteristics make it easier or more difficult for that group to develop sharing arrangements among members. Relatively small groups, groups with members with similar goals, capacities, and histories associated with the harvesting of a resource tend to have an easier time agreeing to the means of distributing flows of benefits. Inversely, large groups or groups with members that have dissimilar goals, varying capital requirements, and varying histories with the harvesting of a resource tend to have a harder time developing resource-sharing agreements among members.

When group allocations are indeed synonymous with the desire to forge co-management arrangements, one of the main questions for policymakers is the degree to which they should involve themselves in the internal group negotiations over resource sharing. In some instances, policymakers have had to decide how resources or benefits would be shared among group members initially. After this initial decision, group members have been free to deviate from the original allocation. Two examples are the catcher vessel segments of the Bering Sea Pollock fishery and the Pacific whiting fishery. In these cases, policymakers determined allocations to individual fishing entities but required that these entities join a harvesting cooperative (a group of vessels harvesting a fishery resource that are jointly liable for certain catch and bycatch standards) to gain access to these allocations. After forming and joining these cooperatives, members subsequently traded allocations among themselves while simultaneously fishing in a rational, profitable manner.

In the case of the Pacific whiting catcher vessel fishery, members had attempted to form a cooperative and reach an agreement about how to share harvesting benefits, but they were unsuccessful and thus fished competitively until the establishment of individual allocations. It was the engagement of policymakers in the internal group-allocation decisions that made the formation of the group—and ultimately the rational fishing activity associated with that group—possible. Alternatively, the catcher-processor segments of the Pollock fishery and Pacific whiting fishery did not need policymakers to help determine individual allocations within these groups. Because of the high degree of similarity among harvesting entities, members were able to agree relatively easily to resource sharing by dividing the group allocation into individual allocations without policymaker involvement.

At the end of the day, there is probably no case where group allocations truly exist. Instead it is more accurate to consider group allocations as a matter of perspective. Policymakers may view their role as allocating to a group, but when considering the set of decisions to be made further downstream, individual allocations may be determined by another set of decision-makers or stakeholders.

When contemplating group allocations it is probably most productive to ask how policy decisions can help foster well performing co-management entities. One critical place for policymakers to consider engaging is the degree to which they should involve themselves in group negotiations over benefit sharing. Depending on the likelihood that the group will coalesce around a resource or benefit-sharing formula, policymakers seeking to create effective co-management entities may or may not want to involve themselves in a way that helps determine such internal-allocation decisions.

8 Grandfathering by Merit

Christopher Costello (U.C. Santa Barbara, NBER, and PERC), Charles Figuières (Aix-Marseille University), and Corbett Grainger (University of Wisconsin, Madison)

Introduction

Environmental markets are increasingly implemented around the world to overcome the tragedy of the commons and to encourage more efficient natural resource use. For example, prominent environmental markets are now widely used to manage air quality, fisheries, biodiversity, ecosystem services, water quality, groundwater extraction, hunting, land, and other natural resources. It has long been argued that the efficient use of these resources depends only on the extraction cap that is set and not on the allocation of rights. But a growing literature suggests that the approach to allocation can significantly affect distributional outcomes, likelihood of program adoption, and even compliance and efficiency once the program is in place (Anderson et al. 2011, Grainger and Costello 2016, Leibbrandt and Lynham 2018).

In practice, rights in environmental markets have been allocated in a variety of ways including by auction, by historical use (which is colloquially called "grandfathering"), to communities, in equal shares, among other formulae. But because economists have historically held that economic efficiency is unaffected by this allocation, economists have largely been silent on the issue of how to allocate rights.

In applied settings, the allocation of rights has become perhaps the most contentious feature of environmental market design. After all, the allocation of rights is effectively a lump-sum allocation of rents. And because the adoption of an environmental market almost always requires some level of buy-in from diverse stakeholders, one allocation formula may be acceptable, while another is outright rejected. This suggests that the allocation of rights can indeed have an enormous effect on efficiency of an environmental market: If rights are allocated in a way that is palatable to the relevant stakeholders, then the market can proceed (and the well-known efficiency gains can be achieved). If rights are allocated in a way that is rejected by stakeholders, then the market institution may be rejected, and the efficiency gains cannot be realized.

What can economics tell us about the political acceptability of any given allocation formula? We take as a point of departure that there are two broad classes of stakeholders whose preferences must be accounted for in the allocation formula. First are the incumbent resource extractors. If the allocation to this group is sufficiently low, they can actually be made worse off under the new market than under the status quo, and may thus rationally oppose the transition. And second, because the resources being governed are typically held in the public trust, the public at large often has a say in the allocation formula. Community advocates, environmentalists, and others argue for allocation based on social or environmental criteria, which we will loosely refer to as "merit." These two interest groups are often in conflict with one another, and so a tension is created between allocation based on (1) historical use by incumbents and (2) merit.

The Case for Grandfathering Based on Historical Extraction

A recent paper by Grainger and Costello (2016) tackles the first approach. That paper develops a model of fishermen with heterogeneous skill who compete in a race to fish, which characterizes the governance regime in the absence of an environmental market. Those fishermen each earn inframarginal rents in proportion to their skill—the intuition is that a high-skill fisherman can outcompete a low-skill fisherman and is thus more successful in the race-to-fish and earns a higher rent. Fish catch is given by a linear harvest technology that depends on fisherman-specific skill, season length, and fish abundance. That theory posits that any given fisherman will support the transition to the environmental market if and only if her overall rent is larger under the environmental market than it is under the race to fish. As argued above, the rent to any given fisherman will clearly depend on the free allocation of rights. (This is of course true whether the fisherman turns out to be a buyer or a seller of rights.)

Under the model described above, it is straightforward to derive an expression for the change in rent to any fisherman that arises from the transition to the environmental market as a function of the allocation of rights. It depends on: (1) the inframarginal rent that can be expected, after trading, under the environmental market, (2) the value of freely allocated rights to the fisherman, and (3) the rents under the limited entry race to fish. Using this model, Grainger and Costello (2016) show that under reasonable assumptions, a fisherman benefits from the transition to the market if and only if she is allocated rights equivalent to at least half of her historical catch (their Proposition 4).

This result may help explain why rights in nearly all environmental markets are allocated by complete grandfathering and are almost never auctioned. While this result addresses the direct economic incentives of fishermen to endorse or oppose the transition, it does not address other stakeholders' preferences, which mainly revolve around concepts of equity or merit in the allocation of rights.

The Case for Grandfathering Based on "Merit"

Environmentally-minded stakeholders often oppose grandfathering based only on historical harvest. They argue that it rewards excessive historical harvests, which are often propped up by illicit behavior, habitat destruction, and other unsustainable practices. Thus, it is argued that grandfathering based on historical harvest rewards the least meritorious extractors. Instead, allocating explicitly based on historical merit would reward sustainable behavior and would thus increase the palatability of the environmental market to these socially and environmentally-minded constituents.

While allocating based on historical harvest will appeal to incumbent fishermen, and allocating based on historical merit will appeal to society at large, these objectives are often in conflict with one another. This raises the question: Is it possible to allocate rights based on both historical harvest and merit, thus satisfying a much larger constituency?

Empirical Example from Global Fisheries

Suppose we have an exogenously defined "merit function" that can be applied across all fishermen in a fishery: $M(x_1,\ldots,x_K)$ which is a function of K observable variables. For example, these could be variables such as the number of hours fishing in marine reserves or whether the fisherman has fished illegally.

Here we provide an illustration of one possible approach to defining merit for fishermen on the high seas. Using Global Fishing Watch (Kroodsma 2018), we identified all fishing vessels that spent at least 5 percent of their fishing effort in the high seas in 2014, 2015, and 2016. There are a total of 4,039 such vessels. For each vessel, we estimated their catch over the three-year period. (This will be the basis for grandfathering rights based only on harvest, without regard for merit.) For each vessel we have also calculated the following variables that relate to merit: flag nation, vessel length, tonnage and power, number of hours fished, number of hours fished in foreign exclusive economic zones (the variable EEZ below), number of hours fished in no-take marine protected areas, number of hours fished in an ecologically sensitive area (the variable $ECOL$ below), tons of CO_2 emitted (the variable $CO2$ below), fishing gear, and number of transshipment encounters.

Using these data, an example of a "merit function" might be:

$$M_i = \alpha_0 - \alpha_1 EEZ_i - \alpha_2 ECOL_i - \alpha_3 CO2_i$$

We can think of each coefficient as the marginal social cost of one unit of the bad activity. For the purposes of this illustration, we use:

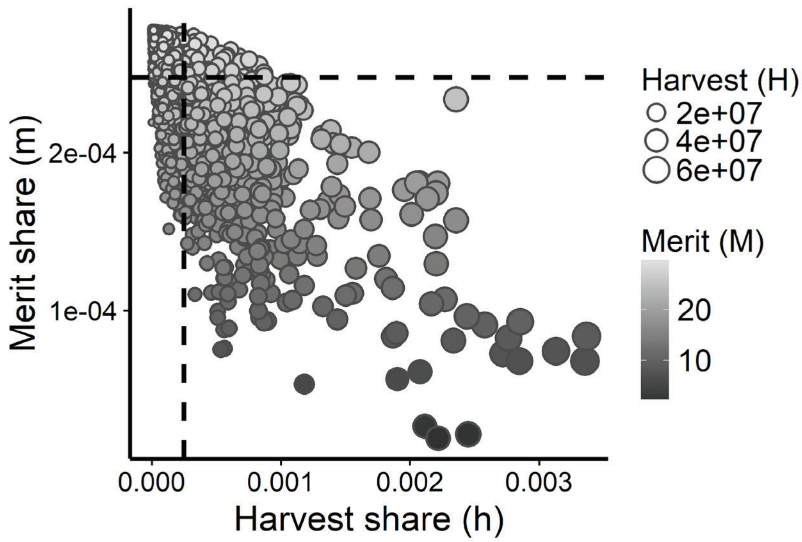
$$\alpha_0 = 30,\ \alpha_1 = 1/1000,\ \alpha_2 = 1/2000,\ \alpha_3 = 1/4000$$

Using this formula, each of the 4,039 vessels has a historical harvest and a historical merit. Scaling each by the total across all fishermen gives the historical harvest fraction (m_i) and the historical merit fraction (h_i). We posit an allocation of rights as a weighted combination of these variables as follows:

$$Allocation_i = \beta h_i + (1-\beta) m_i$$

The parameter β is a policy parameter that determines the weight placed on harvest versus merit in the allocation formula. If we choose $\beta = 1$ then we are allocating purely based on historical catch. (This is the typical "grandfathering" approach in fisheries, water use, and other resources.) If we choose $\beta = 0$ then we are allocating purely based on historical merit. Figure 1 shows the historical harvest share vs. historical merit share for each high-seas fisherman where size indicates raw harvest and shading indicates raw merit (dashed lines indicate mean values). The figure reveals that harvest shares and merit shares indeed tend to be negatively correlated, so allocating primarily based on historical harvest (i.e., for $\beta = 1$) will generally disadvantage those with high merit, and allocating primarily based on merit ($\beta = 0$) will generally disadvantage those with high historical harvest. Note

Figure 1: Historical Harvest vs. Historical Merit



also that there is considerable spread, so many fishermen that have high historical harvest also have high merit, and many fishermen who did not fish much (so have low harvest) did so in very destructive ways so they have low merit.

How will different allocation formulae (i.e., different values of β) advantage or disadvantage individual fishermen (noting that it is straightforward to see that any value of β will result in exactly 100 percent of the harvest rights being allocated across the entire fleet)? Harkening the main result of Grainger and Costello (2016), we are interested in how different values of β affect the share of freely grandfathered rights to each fisherman, and in particular, whether that share exceeds 50 percent of her historical catch.

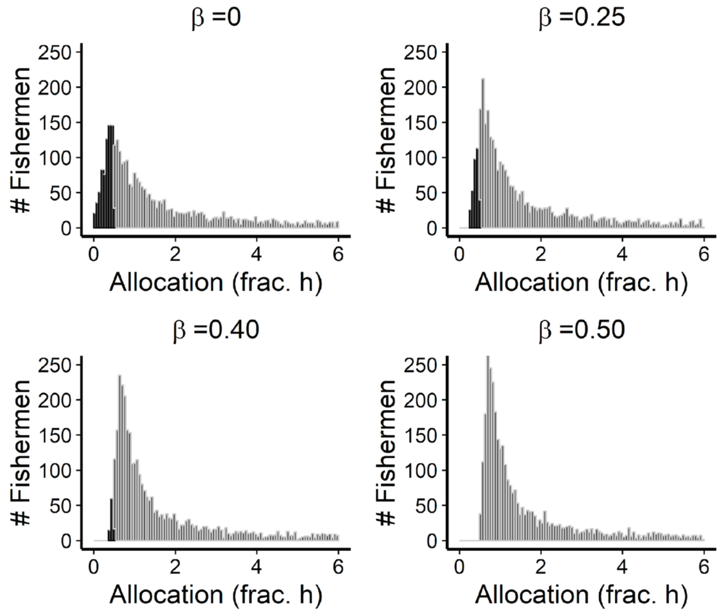
In what follows we illustrate the distributional consequences of different allocation formulae and determine whether that allocation approach is Pareto improving for all fishermen, and if not, for what percentage of fishermen it is disadvantageous.

It is straightforward to manipulate the above allocation equations to define a critical value of β for each fisherman (call it β_i^{crit}), such that fisherman i will be better off under environmental markets if and only if $\beta > \beta_i^{crit}$). That value is given by:

$$\beta_i^{crit} = \frac{.5h_i - m_i}{h_i - m_i}$$

Figure 2 displays a series of histograms of the allocation across all fishermen (as a fraction of h), for four different values of β. Consider the case when allocation is based entirely on merit, so no weight is placed on historical harvest (this is the case where $\beta = 0$). In that case, most fishermen are still better off under the environmental market than they would have been under the race to fish. The shaded area represents the fishermen for whom the free allocation is lower than 50 percent of their historical harvest. When $\beta = 0$, this amounts to 18 percent of the fleet. When $\beta = 0.25$, this shrinks to just 8 percent of the fleet. And for any $\beta > 0.50$, the transition to the environmental market is strictly Pareto improving for all fishermen.

Figure 2: Various Allocations Across All Fishermen



Discussion

As natural resources such as fisheries, forests, water, and biodiversity are overextracted in a common-pool race, environmental markets are increasingly seen as a viable and efficient solution. Yet implementing these markets remains contentious, primarily because the initial allocation of rights creates winners and losers. Incumbent resource extractors argue that without sufficiently large free allocations, they will be made worse off under the environmental market. And the public at large argues that the allocation should be to the most meritorious, not necessarily to those with the largest historical harvest. Whatever the definition, environmental merit is likely to be negatively correlated with historical extraction, which raises a concern: If we allocate purely based on historical use, which is almost always the default approach, then we disadvantage the most meritorious, but if we allocate purely based on merit, then we disadvantage incumbent extractors. Either outcome may be politically unviable, which raises overall concerns about efficiency if the transition to the market is blocked.

In this paper we propose the idea of grandfathering rights in an environmental market based *partly* on historical use and *partly* on merit. We showed that the relative weight of each determines which users are better off, and which users are worse off, under the transition to the market than they would have been under the common-pool race. In our application to 4,039 high-seas fishermen, we found that even if the allocation was entirely based on merit, still 88 percent of fishermen would be better off under the market than under the status quo. And if the allocation was based 40 percent on merit and 60 percent on historical catch, then 97 percent of fishermen would be better off under the new market regime. Results from both our theoretical model and our empirical example suggest that allocating rights *at least partly* based on merit could achieve the dual goals of satisfying constituents who favor merit as a criterion for allocation of public trust resource access and ensuring Pareto improving outcomes for nearly all incumbent resource extractors, all while enabling an overall efficiency gain from the transition to the environmental market.

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RESPONSE

Nicole Sarto (Environmental Defense Fund)

It seems that the authors' objective in developing this grandfathering-by-merit concept is to find a new way to get stakeholder buy-in to allocation and market-based systems when catch history isn't sufficient or aligned with stakeholders' goals and values. In that case, using catch history as the basis for allocation could cause the stakeholders to reject the allocation decision and the market-based system altogether.

The authors discuss equity, and stakeholder perceptions of whether allocations are equitable, as the rationale for using a merit-based approach to allocation. Equity is, of course, critically important to generate stakeholder buy-in to the system, and that is important for the durability of the system. It is important to note, however, that how stakeholders perceive equity can be nuanced and varied across contexts. Equity should not be equated with "merit."

If we consider what would happen in a fishery that applies this merit-based approach, there are a number of ways that the authors' objective may not be realized. Depending on how the stakeholders or fishery managers define merit, large operators, who may have been acting perfectly rationally under the existing regulations and incentives, could be penalized and pushed out of business. From their perspective, this merit-based approach was therefore not equitable, and you would be unlikely to get their buy-in. If they have a lot of political power in the system, as is common, they could effectively prevent the implementation of the market. Perhaps there are better ways to think about how to improve buy-in to a market-based system while not relying entirely on catch history.

An assumption that is present in the paper is that buy-in is always generated by making a majority of people better off than under the open-access system. Surely, the specific players who are better off and worse off has an impact on how the system will be received and implemented. In the absence of an inclusive, participatory process that fairly considers the needs of all stakeholders, those with more political power and representation will need to be the most bought-in for the system to work. Furthermore, this approach relies on the assumption that the least meritorious actors, who will be most penalized by this approach, will comply with their allocation and other regulations associated with the new management system.

There are other ways to generate stakeholder buy-in to a difficult transition to a market-based system. These align closely with the principles of good governance. Think about it from the perspective of the stakeholders who are participating in the process: They want to be treated fairly and to have their goals and values incorporated into the allocation approach. The process should be goal-driven, be inclusive and participatory, weigh the needs and values of all stakeholders, foster transparency, include mechanisms for procedural justice and conflict resolution, embody the principles of good governance and management design, and enable bottom-up decision-making. Putting the resources in the hands of stakeholders that will enable them to design their own allocation mechanism that aligns with their goals and values, whether it be merit-based or otherwise, is likely the best way to generate buy-in.

It is also important to consider how this approach will work in diverse fishery contexts and offer the appropriate caveats and bounds of its usefulness. The context modeled in the paper is an industrial, commercial, international fishery, with actors that are assumed to be operating in highly individualistic and capitalistic societies. If this solution works best in that context, pointing that out would add value and clarity. If the authors believe this solution could work in other contexts, they should discuss those contexts and how the approach could be amended or adapted accordingly to ensure it meets their objective of equity and buy-in.

Many Pacific Island cultures, for example, emphasize the importance of participation in perceptions of equity. The only way work gets done is through talking with stakeholders, in person, and there is immense value placed on everyone feeling included and heard in the decision-making process. In other contexts, like Japan, the needs of the collective are prioritized over the needs of the individual. Fishing cooperatives have different ways of allocating fishing opportunities, but all revolve around the Japanese perception that equal is equitable. Considering these contexts, and others that are dissimilar from the one assumed in the paper, will help the authors think about the constraints or limitations of this "merit" approach, and why it might be important to find other ways to achieve the overarching objective.

9 Leakage and Industrial Subsidies

Matthew Zaragoza-Watkins (Vanderbilt University)

When product markets impose external costs, government regulation can be justified as a means of increasing social welfare. The contribution of anthropogenic greenhouse gas (GHG) emissions to global climate change is a key example. Among economists, there is near consensus that a market-based approach (i.e., cap and trade or tax) should be adopted to correct this critical environmental externality. While a binding global framework for controlling GHGs remains elusive, regional market-based policies have begun to emerge. However, because many emissions-intensive products are traded across borders, and because regulators are typically only able to enforce policies on local facilities, there is a risk that emission reductions at regulated facilities will be offset by emission increases at unregulated facilities, undermining the environmental effectiveness of these policies. Thus, a key challenge for policymakers is how to mitigate the risk of emission "leakage" due to incomplete regulation.

Economists have established that when regulation is incomplete the preferred approach is to complement the market-based policy with a series of border tax adjustments (Fischer and Fox 2012).[1] However, these have proven difficult to implement for a variety of political and technical reasons. Instead, regulators have frequently chosen to provide trade-exposed firms with output-based subsidies in the form of freely allocated pollution permits.

Output-based subsidies can effectively mitigate the risk of emissions leakage, but there are tradeoffs. Subsidizing output can significantly suppress pass-through of permit prices to consumers, eroding the incentive for consumers to substitute toward less emissions-intensive goods. Under a binding cap, this will also push up permit prices, inducing more costly emission reductions from other capped sources.[2] Output-based subsidies also reduce the share of permit value that can be directed to other desirable purposes.[3] Because the opportunity cost and distributional consequences of improperly subsidizing firms to mitigate emissions leakage can be significant, it is important that output-based subsidies are carefully calibrated.

Regulators face significant information problems when attempting to set output-based subsidies. Even within industries, emission rates and abatement opportunities can vary widely from one facility to the next.[4] Thus, the joint impacts of adjustment costs (i.e., cost increases due to induced input substitution) and compliance costs (i.e., the cost of procuring pollution permits) on facilities' marginal production costs are likely to vary significantly within and across industries. Moreover, many of the homogeneous product markets for which subsidies must be designed are imperfectly competitive, with firms charging variable markups. It has been shown that when imperfectly competitive firms face heterogeneous cost shocks, the impacts on total output and welfare tend to depend on changes in the mean and variance of marginal costs, respectively (Février and Linnemer 2004). Thus, setting an optimal output-based subsidy requires the regulator to know (or otherwise infer) the distribution of adjustment costs, compliance costs, resultant marginal costs, as well as the nature of competition among the set of regulated and unregulated facilities in each industry or product market. Despite the prevalence of this policy approach, a normative framework for setting output-based subsidies capable of incorporating technical substitution and imperfect competition has not emerged. In practice, regulators often rely on very limited, static information to set output-based subsidies.

In California, for example, where there is an economy-wide GHG cap-and-trade program covering more than 85 percent of stationary-source emissions, most regulated manufacturers produce goods that are regionally or globally traded. To "minimize leakage," regulated manufacturing facilities are freely allocated GHG permits on the basis of their output and a product-based emissions "benchmark": an emission rate per unit of output set to either 90 percent of the average emission rate among regulated facilities producing that good or to the best-in-class emission rate among those facilities if none has an emission rate at or below 90 percent of average.[5] At the outset of this market, more than 30 percent of emissions permits were freely allocated via this approach, a level that has largely remained unchanged. At that time, the California Air Resources Board acknowledged that little was known about the distribution of potential abatement opportunities within and across industries and that staff would need to closely monitor industry for signs of leakage.[6]

In a new study, I consider two questions that are motivated by California's policy experiment (Zaragoza-Watkins 2018). When product markets are imperfectly competitive and firms can reduce emissions via technical substitution, how should policymakers optimally set output-based subsidies to target an efficient level of emissions leakage? And, in practice, how have the incentives created by California's GHG cap-and-trade program design impacted emissions and output at regulated manufacturing facilities?

I begin the analysis by developing a theoretical framework in which a number of producers with heterogeneous constant marginal costs and abatement opportunities compete in quantities. I find that the partial-equilibrium net effect of the permit price and output-based subsidy on aggregate output can be characterized by the change in average marginal costs. I show that to avoid emissions leakage, the regulator's objective is to set a schedule of output subsidies that will leave the sum of industry-wide output unchanged. If firms compete in quantities, then the equilibrium condition simply requires that the output subsidy be calibrated to exactly offset the average marginal compliance cost associated with the permit price. One implication of this analysis is that the combination of permit price and output subsidy can increase or decrease the total social cost of production among regulated firms, depending on whether the permit price increases or decreases the variance of marginal costs of production. I then derive a formula for the optimal level of output-based subsidy, which maximizes welfare by equating the marginal social cost of leakage to the marginal cost of public funds (i.e., the opportunity cost of using allowance value for leakage mitigation).

Next, I evaluate the effects of California's GHG cap-and-trade market design on emissions, production, and emissions intensities among covered manufacturing facilities. To construct my counterfactual, I collect detailed information from more than 3,000 manufacturing facilities located across the country from the U.S. Environmental Protection Agency's Greenhouse Gas Reporting Program and Toxic Release Inventory. I combine the data into a facility-level panel of emissions, output, and emissions intensities from 2010-16. To estimate the combined causal effects of the cap-and-trade permit price and output-based subsidies, I compare changes in outcomes at treated facilities to changes in outcomes at facilities in other states that would have received output-based subsidies under California's cap-and-trade program, had they been located in California. In particular, I match each treated facility to a set of counterfactual facilities located outside of California that operate within the same 6-digit North American Industrial Classification System codes (i.e., the same product market), and that are similar on observable characteristics, and employ a matched panel fixed-effects design to estimate the average treatment effects of the policy on the growth rates of emissions, output, and emissions intensity of treated facilities. To my knowledge, this analysis provides the first empirical evidence on the effectiveness of output-based subsidies in the context of a cap-and-trade program.

The California program provides a unique opportunity to study the impact of output-based subsidies. A primary challenge for evaluating any non-experimental program is identifying a valid counterfactual. This has proven to be especially challenging for analysts wishing to study market-based mechanisms in which all market participants tend to be directly or indirectly treated, violating the stable-unit treatment value assumption (SUTVA). For example, SUTVA would be violated if competition among treated and counterfactual plants implied that policy-induced output changes at treated plants are offset by opposite-signed output changes at counterfactual plants. If this were the case, including the indirectly treated plants in the set of counterfactuals would bias causal estimates of the effect of the policy on output away from zero. In the context of California's policy, on the other hand, SUTVA may not be violated if the output-based allocation policy effectively mitigates the sorts of output spillovers that would be induced by cost increases due to the permit price by suppressing cost pass-through. To investigate the marginal impacts of the policy on output and emissions intensity (i.e., the two components of emissions), I decompose emissions changes into changes in output and emissions intensity. If output-based subsidies are effective, theory predicts that the primary adjustment margin will be emissions intensity, while output will remain relatively unchanged. Thus, somewhat crucially, the decomposition of emissions changes into output and intensity allows for a careful (albeit indirect) analysis of the SUTVA assumption.

My results indicate that emissions at treated facilities have fallen sharply, driven primarily by improvements in emissions intensity, while production has been essentially unaffected. I investigate the plausibility of SUTVA in several robustness checks, and find no evidence of any systematic violations, consistent with the effective use of output-based subsidies. Next, I explore heterogeneous treatment effects and find evidence of significant within- and across-industry heterogeneity among treated facilities. Consistent with the predictions of my theoretical framework, within-industry heterogeneous treatment effects (i.e., among treated facilities) appear to be driven by shifts in output favoring treated facilities with relatively lower ex post emissions intensities. Across industry heterogeneity suggests that levels of output-based subsidies may be averagely correct (in the sense of mitigating average adjustment and compliance costs) across industries, but that significant between-industry differences persist in the ability to substitute away from emissions-intensive inputs.

In conclusion, I provide evidence that output-based subsidies can be an effective tool for achieving emissions reductions from capped sources while simultaneously mitigating the risk of emissions leakage. I also show that the sensitivity of particular facilities to the level of subsidy can vary significantly within and across industries, depending on the competitive position of that facility or set of facilities, relative to the composition and organization of the industry, and in particular the level of competition from uncapped sources. Regulators should exercise care in setting output-based subsidy schedules.

Endnotes

1 Border tax adjustments are taxes on imported goods (and subsidies to exports) designed to approximate the incentives created by a GHG tax or cap-and-trade permit price. Accordingly, imports are taxed according to their embodied GHG emissions. Imposing similar compliance obligations on imported goods can create a level playing field between imports and domestically produced goods. However, due to the complexity of supply chains for many finished goods, accurately quantifying embodied emissions can be prohibitively difficult.

2 This phenomenon does not arise under emission taxes or rate-based standards, because the level of the tax or standard is not a function of total regulated emissions.

3 Examples of other desirable uses of permit value (or revenue) include reducing distortionary taxes, providing lump-sum compensation to consumers for the higher cost of goods and services or investments in cost-effective emissions reductions from uncapped sources.

4 Indeed, the gains from trade that contribute to the efficiency of market-based policies partly derive from heterogeneous abatement costs. The other most commonly noted advantage is the opportunity to induce regulated facilities to undertake cost-effective emissions reductions that they might not otherwise choose to undertake in response to command-and-control regulation.

5 Assembly Bill 32, the piece of legislation authorizing the establishment of a cap-and-trade program, directs the California Air Resources Board (CARB), the implementing agency, to "minimize leakage." Taking as given that California's objective is to avoid any leakage, each benchmark would suffice to describe CARB's expected difference between *ex ante* and *ex post* average marginal production costs (i.e., net of adjustment and compliance costs) among regulated facilities.

6 See Appendix B to the Initial Statement of Reasons supporting the July 2011 Cap-and-Trade Regulation, available at https://www.arb.ca.gov/regact/2010/capandtrade10/candtappb.pdf.

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RESPONSE

Daniel Kaffine (University of Colorado Boulder)

While environmental markets have been proposed and implemented in a variety of settings, a frequently raised and important concern is “leakage” due to incomplete regulation. That is, because the environmental market does not fully span the entire market (either in terms of spatial jurisdiction, goods, sectors, firms, or participants), activities formerly undertaken in areas regulated by the environmental market may “leak” to unregulated areas. In “Leakage and Industrial Subsidies,” Matthew Zaragoza-Watkins theoretically and empirically examines output-based allocation, a policy implemented to mitigate emissions leakage in California’s greenhouse gas (GHG) cap-and-trade program.

Output-based allocations can be thought of as subsidies to firms for every unit of output they produce—in practice, this was accomplished via freely allocated permits to emissions-intensive and trade-exposed firms. There are both efficiency and distributional considerations associated with output-based allocations. On the efficiency side, reductions in global pollutants from regulated firms that are offset by leakage to unregulated firms yield no benefits—a ton of GHG emissions in Arizona is just as harmful as a ton emitted in California. On the distributional side, firms that are emissions-intensive and face export competition via trade bear a large burden under carbon prices, and leakage may result in loss of market share or employment for those firms and industries if production simply relocates out of state.

Zaragoza-Watkins first considers the theoretical question: How should regulators set an output-based subsidy to mitigate emissions leakage? Using a tractable, analytical framework, he shows that the per unit subsidy should be equal to the average marginal compliance cost in a given industry. Intuitively, leakage occurs because the carbon market price raises the average marginal cost of production across firms in an industry, reducing output in the regulated industry but leading production to shift elsewhere. By offering an output subsidy exactly equal to that increased average compliance cost, the incentive to reduce output is removed, as is the associated leakage. Importantly however, because firms still face a price on carbon, they retain an incentive to reduce emissions via other channels such as abatement technologies or input substitution, resulting in “real” reductions in aggregate emissions.

Next, Zaragoza-Watkins examines how the actual output-based allocations policy in California has performed in practice. To identify the causal effects, outcomes at regulated manufacturing firms in California, which received output-based allocations and are subject to cap-and-trade, are compared to similar firms located in other states, which did not receive output-based allocations since they are not subject to cap-and-trade. Specific outcomes examined are total emissions, emissions intensity, and production—reductions in production at California firms would be indicative of leakage. While preliminary, estimates show that emissions fell by 7 percent on average at California manufacturing facilities subject to cap-and-trade and output-based allocations, consistent with the general policy goal of reducing GHG emissions. Importantly, this reduction in emissions is driven by a reduction in emissions intensity and not by reductions in production.

On average then, the empirical estimates suggest the output-based allocation scheme implemented by California succeeded in the goal of mitigating leakage. However, preliminary evidence suggests that focusing on the average amount of leakage overlooks significant heterogeneity within and across industries in terms of firm response. Within an industry, production shifts from high emission-intensity firms to low emission-intensity firms, as expected. Across industries, the story is even more interesting. The actual output-based allocations policy implemented used a series of emissions benchmarks to determine the allocation to each industry, in contrast to Zaragoza-Watkins' proposed theoretical allocation based on compliance costs with cap-and-trade. This would suggest that industries with low compliance costs that could easily substitute away from emissions-intensive inputs may have received a larger subsidy than needed to avoid leakage, while industries with high compliance costs due to limited substitution options may have received an insufficient subsidy to avoid leakage—preliminary estimates suggest that this may be the case.

There are many interesting directions for further theoretical and empirical research in this area. Continuing to analyze heterogeneous responses to this policy by firms and industries may yield additional insights into the distributional and efficiency aspects of cap-and-trade with output-based allocations. Such analysis may also help isolate the effects of output-based allocations as distinct from the currently estimated combined effects of the allocations and cap-and-trade. Follow-up research questions to be considered include: How might output-based allocations be used in other environmental markets—for example, is there scope for such schemes in marine-resource or land markets? How does an output-based allocations policy interact with permit prices, and might it impact other distributional goals such as auction revenue allocation? And finally, what is the socially optimal level of output-based allocations? While California mandated that leakage be minimized, there are both costs and benefits of output-based allocations, and "zero leakage" is likely suboptimal. Zaragoza-Watkins' work provides a useful analytical framework and preliminary empirical estimates that move us in the direction of understanding what an efficient output-based allocations policy under the threat of leakage might look like, as well as the distributional implications that follow from it.

10 Auctioning Payment for Ecosystem Contracts

B. Kelsey Jack (U.C. Santa Barbara)

Payments for ecosystem services (PES) are a type of conservation program in which individuals are compensated for providing ecosystem or environmental services.[1] For example, to promote forest cover, a PES program might pay landholders to leave forest intact or plant new trees. PES programs are popular in low-income countries where requiring landholders to conserve without compensation could exacerbate poverty.

A core principle of PES programs is that they are voluntary: A payment is offered for some environmental outcome, and a landholder chooses whether to participate. The voluntary nature of PES makes self-selection into a program one of the key factors determining its environmental benefits and cost effectiveness.

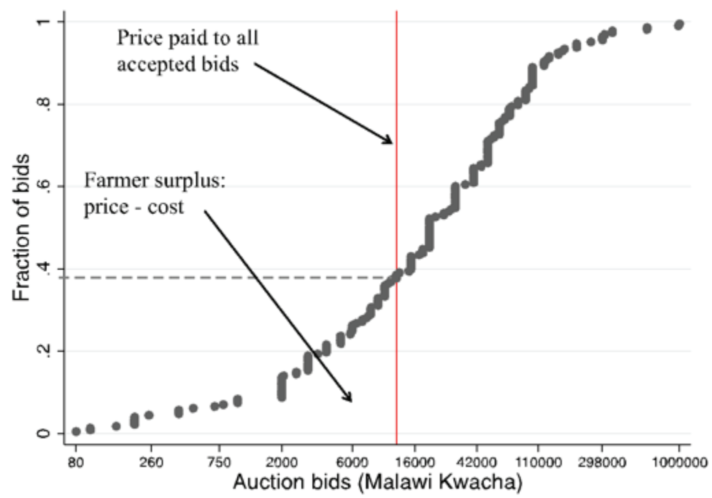
Rules for program enrollment determine which landholders receive contracts. At any given contract price, only landholders whose costs of participation are covered by the price will choose to enroll. A price that is too high will lead to too much supply by eligible landholders; too low a price will have the opposite effect. Thus, getting the price right, and allocating contracts to landholders able to supply the environmental service at the lowest cost is important for a PES program's cost effectiveness, or conservation achieved per dollar spent.

Auctions have been proposed as an approach to overcoming the fact that landholders tend to know more than the PES program designer about the price that will make them willing to enter into a PES contract (and comply with it).[2] One benefit of an auction is that it determines the contract price based on the supply of landholders willing to provide environmental services at that price and simultaneously allocates contracts to these landholders. Auctions have been used to allocate PES contracts in the United States and Europe in large-scale programs and in developing country programs on a smaller scale.

The example discussed here is from Malawi and involved an active PES contract that requires tree planting, which landholders were unlikely to undertake absent the contract. The auction format was designed to make "truthful bidding" the dominant strategy for landholders. In other words, landholders could do no better than to tell the truth about the minimum price that would make them willing to enroll.

The auction proceeded as follows: Eligible landholders submitted sealed bids listing their minimum contract price. Under a fixed budget available to pay for conservation, bids were accepted, starting with the lowest, until paying all accepted bids the value of the first rejected bid exhausted the budget. As a result, all enrolled landholders received a price above their minimum willingness to accept. Paying above a landholder's bid is necessary to induce truth telling. Paying all winning bids the same price provides an extra transfer to enrolled landholders—the difference between their bids and the amount they are paid—but avoids potential logistical complications and fairness concerns associated with offering different prices to different landholders. Figure 1 summarizes the auction results.

Figure 1. Farmer bids in Malawi auction.

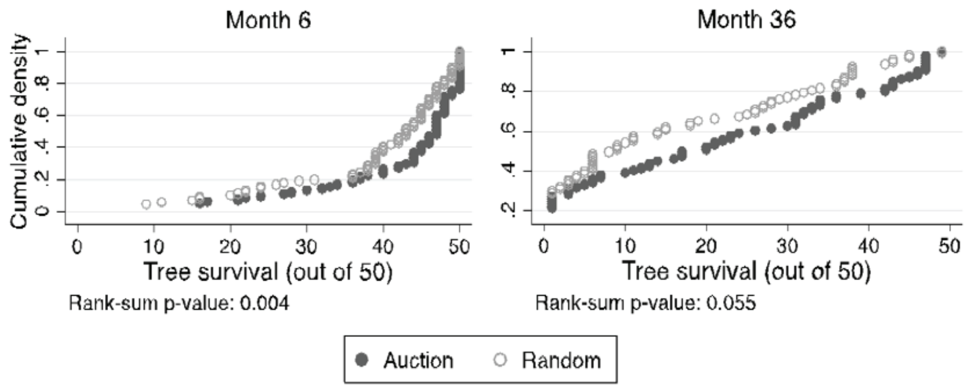


Source: Jack (2013).

The study described in Jack (2013) used a randomized controlled trial to test the impacts of allocating PES contracts through an auction. Half of eligible farmers were assigned to the auction and the other half to a lottery, which assigned contracts at random. Regardless of the allocation treatment, all contracted farmers received the same contract price and terms (a piece rate per surviving tree paid in installments for up to three years). Results showed that tree survival was significantly higher among farmers who received the contract through the auction, consistent with better targeting of contracts to low-cost landholders through the auction. Figure 2 summarizes tree survival outcomes after 6 months and 3 years.

While these results show that auctions have the potential to increase the cost effectiveness of a PES program relative to allocating contracts at random, they say little about distributional equity. The positive transfer embedded in the contract price may raise concerns about whether an auction privileges richer or better-educated landholders. Similarly, if farmers misunderstand the bidding incentives and end up with a contract that does not cover their costs of conservation, they may be made worse off as a result of the contract. While contracts are voluntary, power imbalances between the organizations or governments that offer the contracts and the landholders that receive them may make landholders reluctant to default.

Figure 2. Tree survival by treatment group. CDFs shifted to the right imply higher tree survival.



Source: Jack (2013).

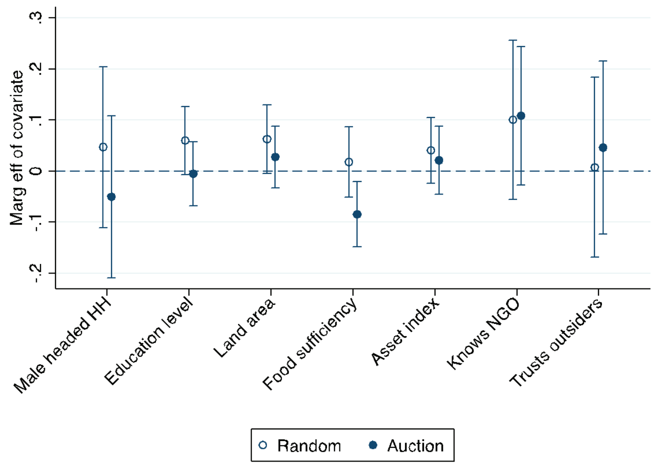
These concerns are legitimate; assessing their importance depends on the viable alternatives. For example, first-come, first-served approaches to contract allocation—which are common—may be more prone to elite capture than an auction, which considers all bids in an objective process. Other potential approaches to allocating PES contracts include selecting among interested landholders based on the environmental benefits their parcels offer or based on other criteria such as poverty levels. While these approaches may explicitly consider factors other than cost of providing environmental services in determining which landholders receive contracts, more complicated auction designs can, in principle, take these factors into consideration in evaluating bids.

This discussion asks two questions about the distributional equity associated with auctions for PES contracts.[3] First, does an auction systematically privilege better-off landholders, and to a greater degree than other approaches to allocation? Specifically, the inclusion of poor or otherwise marginalized landholders will depend on how these characteristics are correlated with conservation costs. This, however, will be true of any PES contract that offers a fixed payment for conservation (i.e., the lowest-cost landholders will tend to enroll). The bidding process used in an auction may make other factors important, such as understanding of the bidding process or trust in the auctioneer. Second, do contracts received through an auction leave landholders worse off than the same contracts allocated through other processes? This might occur if receiving a contract based on a bid increases a perceived obligation to comply even under unfavorable terms.

I use data from the Malawi experiment described above to investigate these questions; many of the results appear elsewhere (see Jack 2013, Jack and Cardona Santos 2017 and Jack and Jayachandran 2018).

Figure 3 summarizes the relative likelihood of receiving a contract through the lottery (at random) versus through the auction, as a function of different farmer characteristics. Specifically, a binary contract indicator is regressed on an interaction between the allocation treatment and the characteristic of interest. Marginal effects of the characteristic, for each treatment group, are plotted below. Continuous variables (education, land area, assets) are normalized to a mean of zero and standard deviation of one. The first five variables are increasing with socioeconomic status. Thus, a higher estimate means that better-off farmers were more likely to receive a contract. For all five poverty proxies, better-off households were less likely to receive a contract through the auction than through the lottery, though differences are not generally statistically significant.

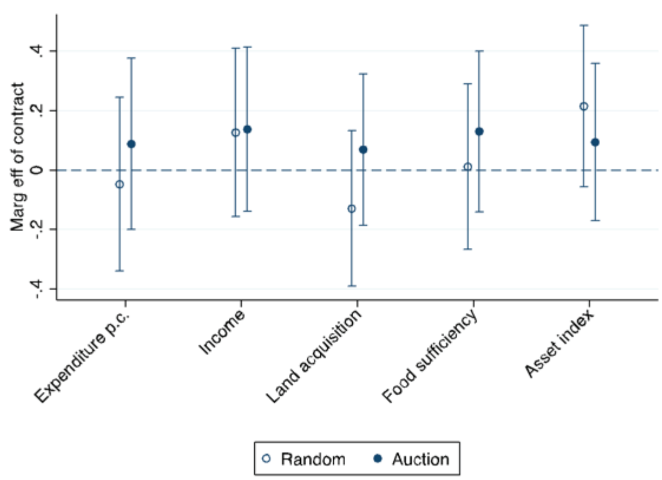
Figure 3. Marginal effect of farmer characteristics on the likelihood of receiving a contract, by treatment group. Continuous variables are normalized to a mean of 0 and standard deviation of 1.



The last two characteristics in Figure 3 examine whether other factors likely to be important for behavior in the auction matter, such as trust in the implementing organization (Jack and Jayachandran 2018). Both experience with the implementing NGO and trust in outsiders result in a slightly higher likelihood of receiving a contract when it is allocated through an auction, though results are again very imprecise.

On the second question, poverty measures were analyzed three years after contract allocation and show slightly positive if modest effects of receiving a contract through an auction, relative to receiving a contract at random, on most outcomes (see also Jack and Cardona Santos 2017). Results are summarized in Figure 4, which plots the coefficient of a regression of the socioeconomic outcome on an interaction of an indicator for the treatment and for whether the farmer received a PES contract. All measures are continuous and normalized to have a mean of zero and a standard deviation of one.

Figure 4. Marginal effect of contract on outcomes. Outcomes are normalized to have a mean of 0 and a standard deviation of 1.



This finding that receiving a contract through the auction led to slightly better socioeconomic outcomes is consistent with the main result (shown in Figure 2) that the auction led to higher tree survival, and presumably greater surplus. Overall, these results show little support for the concern that the auction makes landholders worse off than other means of allocation.

These analyses offer a starting point for assessing the distributional effects of allocating PES contracts through an auction. However, the evidence is extremely limited: from a single, simple auction design involving a particular contract and a narrow set of eligible landholders. Further analysis of distributional impacts of alternative approaches to contract allocation and enrollment of landholders into PES programs can help balance distributional concerns with cost-effectiveness objectives. In addition, investigating other auction and contract designs, as well as alternative approaches to allocation is a promising direction for future research.

Endnotes

1 This summary borrows heavily from Jack and Jayachandran (2018), Jack (2013) and Jack and Cardona Santos (2017), including some direct excerpts.

2 This objective of targeting landholders with a low cost of enrolling is most obvious for PES programs that require actions by the landholder that she would likely not have taken absent the program, such as afforestation or reforestation activities. More passive land use outcomes, such as avoided deforestation tend to require more sophisticated targeting approaches because the inframarginality or additionality problem is worse. In principle, a more sophisticated auction can achieve this goal.

3 General concerns about the distributional equity of PES are beyond the scope of this summary, which focuses on the distributional implications of auctioning PES contracts relative to allocating them in some other way. For example, by contracting with individuals and households with reasonably secure land tenure, PES programs may be regressive regardless of how contracts are allocation. Jayachandran et al. (2017), for example, find some evidence of this in their study of PES impacts in Uganda.

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RESPONSE

James Salzman (U.C. Santa Barbara and UCLA)

An obscure term just fifteen years ago, payments for ecosystem services (PES) have come of age. There are now more than 550 PES programs around the world, with combined annual payments over $36 billion. Governments, businesses, and NGOs have taken a special interest in this approach, as have property-rights advocates who support the contract-based mechanisms found in many PES schemes. Scholars have been equally enthralled, with more than 1,900 PES journal articles in 2016.

These articles have ranged from analyses of how best to structure PES programs and their different payment schemes to assessing the biophysical effectiveness of PES and their economic efficiency. A few of these studies have focused on the social welfare impacts for PES participants, demonstrating neither strong positive nor negative impacts on poverty.

Despite the breadth of studies on the many different aspects of PES, none has carefully considered the distributional impacts. B. Kelsey Jack's article provides an important first foray into this area, suggesting a number of fruitful research questions.

Jack looks at a sliver of the distributional impact question, focusing on reverse auction mechanisms. Employed most notably in the U.S. federal government's Conservation Reserve Program, reverse auctions are particularly effective in cases of information asymmetry. In the case of payments for ecosystem services, landowners hold superior information in terms of what they are willing to receive for PES provision and, often, though not always, the capacity of their lands to provide services.

Jack bases her findings on field work in Malawi, asking two questions. First, does an auction systematically privilege better-off landholders to a greater degree than other approaches to allocation? Second, do contracts received through an auction leave landholders worse off than the same contracts allocated through other processes? She finds little support for either question. These findings suggest that auctions should neither be preferred nor avoided as PES payment mechanisms on distributional grounds.

These are important findings, but Jack's research is, in my view, more significant in suggesting three new research questions:

1) Why would we expect distributive-justice problems with auctions in the first place?
2) Why would we expect *greater* problems with auctions than with other allocation mechanisms?
3) Do program goals matter?

To answer the first question, we need to understand better what drives participant behavior in auctions. Jack notes that factors other than just the cost of conservation appear important for determining bidding in an auction. One could imagine a range of factors that would encourage participation, including positive past interactions with the implementing organization or trust in outsiders.

Equally, and this implicates the second question, other factors might discourage participation in an auction. These include relatively high costs of participation that might keep out poorer landholders and inadequate understanding of bidding strategy or contract terms. Random contracts likely avoid these barriers. Moreover, Jack proposes an additional concern with distributional consequences following an auction: hesitancy of poorer landowners to default on their PES contract.

The third question represents more musing than grounding in experience, but I wonder to what extent the impacts of instrument choice depend on the program goals? It may well be that different types of payment mechanisms—e.g., reverse auctions, offsets, private contracts, subsidies—result in different distributional outcomes. But it may be the case that these differences *also* result from the program goals. Most PES programs seek to provide three goals (sometimes in combination): provision of services, poverty alleviation, and effective spending (ecosystem services value for money). It is worth considering whether the distributional impacts of the different allocation mechanisms may vary based on program goals. One possibility is that distributional equity is least pronounced when the goal is poverty alleviation and most pronounced when the goal is cost-effective service provision.

Answering these three questions is critical to designing better PES schemes. The challenge is how to design field experiments to assess: (1) which factors encourage participation in reverse auctions, (2) the distributional outcomes of other allocation mechanisms relative to auctions, and (3) whether distributional outcomes are related to program goals. Jack's article provides a valuable service, setting forth a clear challenge for social scientists to design their research projects in a manner that isolates these factors so they can be meaningfully examined in the field.

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