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Spillovers from Conservation Programs

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Abstract

Conservation programs have increased significantly, as has the evaluation of their impacts. However, the evaluation of their potential impacts beyond program borders has been scarce. Such spillovers can significantly reduce or increase net impacts. In this review, we discuss how conservation programs might affect outcomes beyond their borders and present some evidence of when they have or have not. We focus on five major channels by which spillovers can arise: (1) input reallocation; (2) market prices; (3) learning; (4) nonpecuniary motivations; and (5) ecological-physical links. We highlight evidence for each channel and emphasize that estimates often may reflect multiple channels. Future research could test for spillovers within different contexts and could separate the effects of different channels.

INTRODUCTION

Conservation programs have increased significantly over time, both in number and extent. For forests, the spatial extent of conservation programs has increased in part due to concern about deforestation and its impact on climate change. This has added to ongoing concerns about threats to the provision by forests of ecosystem services, such as habitat for species, water quality, and the regulation of water flows. Other conservation contexts, such as those concerning water, feature similar threats and conservation responses.

The leading approach to land-based conservation has been the evolving network of protected areas (PAs), within the boundaries of which various types of public actors have restricted or banned activities. Yet policies to stimulate conservation on private lands, such as payments for ecosystem services (PES) to create private incentives, have also increased. Furthermore, combinations of public and private governance are growing. Common examples include various forms of extractive reserves, which are variations on PAs, and the certification of logging concessions by third parties for consumers and public agencies.

Recently, there has been an explosion in the evaluation of impacts from conservation programs, particularly for forests (e.g., Andam et al. 2008; Pfaff et al. 2009, 2015a,b; and Sims 2010 on PAs; Alix-Garcia et al. 2012, Arriagada et al. 2012, and Robalino & Pfaff 2013 on PES; and Ferraro & Pattanayak 2006 and Pfaff & Robalino 2012 for general discussions). Economists have pushed to estimate the (unobservable) counterfactual baselines, i.e., what would have happened in the areas treated by the conservation programs had those programs not existed.

However, whereas estimation of baselines (and thus impacts) has received some rigorous attention, the spillover effects of conservation—on areas outside program boundaries—are often ignored. In other words, the literature has focused on the impacts on the areas covered by programs (the "treated" areas) but for the most part has ignored any impacts on those areas not covered by programs (the "untreated" areas) yet potentially affected by programs through spillovers.

Spillovers from conservation programs are the focus of this review. We distinguish five channels through which spillovers may arise and use this classification to summarize the existing evidence from the empirical literature, acknowledging that multiple channels could often be relevant for any outcome. These are: (Channel 1) input reallocation; (Channel 2) market prices; (Channel 3) learning; (Channel 4) nonpecuniary motivations; and (Channel 5) ecological-physical links. Each channel is considered below.

Overall, we find that interest in and evidence on spillovers are growing. The literature makes clear that spillovers may or may not occur, and if they do, they can vary across contexts in their magnitude and even in their direction. Spillovers through the learning channel, for instance, often go in the direction of the program's goals. When spillovers go in directions counter to programs' goals, how best to avoid them varies with the channel. Empirical identification of spillovers' magnitudes can be a challenge, given the limitations of the available data. However, interest in spillovers is growing sufficiently so that research designs are starting to consider spillovers.

Stepping back from the particular conceptual as well as empirical approaches within the literature, more consistent reporting may provide a better understanding of the scope and scale of spillovers. In terms of ongoing studies, gains also may result from a more empirical focus on causal identification (internal validity), external validity, or more of a sense of variation in spillovers across contexts, as well as on the identification of the channels through which spillovers arise.

As noted above, we organize this review around the aforementioned five channels. Each of the five numbered sections discusses the related empirical literature for the corresponding channel. In each, examples often focus on forests, yet they illustrate and sometimes cite different applications.

Finally, in our Discussion, we summarize key points and consider some challenges and directions for future research.

Channel 1: Input Reallocation

Slippage from restrictions. Profit maximizers facing restrictions on the use of some of their land may alter uses of other parts of their land. Thus, even if a conservation program that imposes some restrictions generates an impact relative to baseline on forest lands that are treated, it could still have no net impact if reoptimization leads to an equal amount of slippage, i.e., above-baseline clearing of other lands owned by the affected landowner.

Consider a restriction that blocks implementation of agricultural plans, such as a new PA that foils crop expansion or a new public requirement that agricultural lands be reforested. Such program mandates could be either uncompensated (e.g., PAs) or compensated (e.g., PES). PES can be implemented as a mandate without truly voluntary choice, albeit with compensation; such a situation may occur in China (Mullan & Kontoleon 2009, Uchida et al. 2007). Yet PES usually function as voluntary restrictions, for example, land-use limits if one signs up to be paid. Given such a newly agreed restriction, a landowner might decide to carry out agricultural activities on land that was previously planned to be in forest.

It is natural to ask why land that could profitably be shifted into an agricultural activity in this way would not already have been used in agriculture within the initial plans, even without the conservation program. One reason is scarce inputs, such as labor and capital. Inputs freed up by the conservation restrictions on the use of some lands can be reallocated, raising profit elsewhere. Thus, given the program, agriculture can outcompete forest on some parcels where it did not do so before, due to falling marginal productivity of inputs or to market failures that limit the supplies of those inputs. For instance, if a farm has only its household labor, and restrictions block the use of labor in one part of the farm's lands, that labor is likely to be used elsewhere.

Although it is often hard to be sure of the exact mechanism at play, there is evidence on the occurrence of such slippage. Much of the evidence relates to the US Conservation Reserve Program (CRP), which sets aside agricultural land for forest regrowth, although the evidence is also relevant for deforestation. Stories have been put forward about input reallocation under the CRP, which is sometimes called the substitution effect. However, the evidence has been contested. Wu (2000) argues that the level of slippage in the CRP may be as high as 14% of the land area under the program. Yet Roberts & Bucholtz (2005, 2006) argue that the identification of slippage is difficult, given the data used in those analyses. Unobservable factors correlated with the CRP treatment may also correlate with the conversion of cropland. Using the erodibility of land as an instrumental variable, which should not be correlated with eligibility for CRP, the authors conclude that there is insufficient evidence of slippage within the CRP.

This debate spurred additional research to improve upon the empirical strategies utilized to identify slippage, such as the use of more disaggregated data and better identification strategies. For example, using satellite imagery that allows more control variables in conjunction with county-level information, Fleming (2010) finds slippage effects of approximately 4% of the land area, which are smaller than those suggested by Wu (2000). Uchida (2014) further improves on the data by compiling a panel at the farm level that can control for farm characteristics, farm-fixed effects, and time-variant county-fixed effects. The nature of the data only allows the identification of in-farm leakage effects, due to the possibility of overly controlling via county-year fixed effects, which will capture the changes in market prices. Using this approach, Uchida (2014) finds high slippage effects and that the farms with a higher share of land not planted in crops have higher slippage effects.

Slippage from resource transfers that compensate for restrictions. Some restrictions on land use in private properties are compensated with payments. The resources provided by such transfers can reduce the constraints upon land use resulting from limited capital. Landowners with credit constraints can expand production on untreated lands using PES funds, or alternatively, keep production constant and buy more goods supplied from other untreated lands.

In a national PES program in Mexico, Alix-Garcia et al. (2012) find that slippage reduces the net impact from quite a large voluntary conservation program. In particular, they note that such impacts are larger in poorer areas, where credit constraints are more likely, and also show that an increase in the number of commercial banks reduces slippage. Other subsidies and cash transfers, such as cost-sharing programs to promote conservation practices, can generate slippage. Lichtenberg & Smith-Ramirez (2011) find that such programs decrease the share of vegetation cover within a farm.

Channel 2: Market Prices

Leakage from direct effects on supplies and demands. A conservation program can shift the supplies of agricultural and forest goods, which in turn, can shift the demands for inputs into the production of those goods, such as labor and capital. Shifts in quantities, if sufficiently large, can shift relative scarcities enough to bring about changes in market prices, which can then affect land use outside of the program area. The price shifts can generate relocations of production to untreated lands (see Armsworth et al. 2006 and Lim et al. 2017 for conceptual discussions, noting that Lambin & Meyfroidt 2011 and Meyfroidt et al. 2013 stress global trade). Here, we focus on such changes in land use by untreated landowners. We define leakage as such shifts by untreated landowners that run counter to program goals.

If land-use restrictions in conservation programs are successful—if they have an impact relative to what would have occurred in the absence of a program—then some treated lands that would have been used for agriculture would instead remain in forest. If such programs are large enough, the supply of agricultural output will decrease enough that agricultural prices will be higher than they would otherwise have been, and the demand for inputs for agriculture will be lower than they would otherwise have been. Higher output and lower input prices generate incentives to change the uses of lands that are not treated by the program (Robalino 2007, Robalino et al. 2015a, Wu 2000). For example, they could encourage an expansion of agriculture on such untreated land.

However, if demand for labor falls enough in the region directly impacted by the conservation program, it could lead to out-migration (Herrera 2015). That makes ambiguous the prediction about the effect of a program on local wages. Sufficient out-migration could also reverse the direction of local spillovers in that it could lead to lower deforestation near PAs (Herrera 2015). Movement of labor also raises the issue of spillovers beyond the forested lands immediately proximate to where the program is implemented. The expected leakage could still occur, yet far from the PA.

Empirical evidence on land-use restrictions of sufficient magnitude to change prices at local levels exists in the PA literature, where we can also see the value of delineating spillover channels. A common empirical strategy in studying local spillovers is to examine whether forest immediately adjacent to PAs is more or less likely to be cleared than it would have been without a PA. In Costa Rica during 1986–1997, for instance, the probabilities of deforestation within the rings immediately around PAs were not significantly different from those for similar lands far from PAs (Andam et al. 2008, Robalino et al. 2015a). However, in light of the specific mechanisms by which such leakage might be encouraged or discouraged, one might want to distinguish between the areas within those rings that are near to roads and those that are far from roads, as well as distinguish between the areas near versus far from park entrances, where tourism is important. Where the profitability of agriculture (relative to forest) is most likely to rise due to the creation of a PA, i.e., far from park entrances but near roads, leakage is indeed found to be significant (Robalino et al. 2015a).

The more general idea here is that spillovers can be heterogeneous across settings. Among other examples (see, e.g., Ferraro et al. 2013), in studying PAs in Madagascar, Desbureaux et al. (2016) suggest that the magnitude of spillovers varies with the density of the local population. This may imply some complicated dynamics, because as previously noted, conservation programs can have effects on migration.

As the migration issue demonstrates, the distances at which spillovers may occur are difficult to assess. Robalino et al. (2015a) show that spillovers vary even across concentric rings around PAs. Clearly, they may vary farther away as well, as programs can affect the uses of distant lands. De Sá et al. (2013) suggest that sugarcane expansion in the state of São Paolo, Brazil, affected land use thousands of miles away within the state of Amazonas because cattle ranching was displaced.

Channels (and net spillover effects) are also likely to differ across types of programs. Yet there is evidence that a very different kind of program from a PA network, such as a logging concession, can generate a similar increase in deforestation outside of its borders, even if it lowers rates of forest clearing internally (Ewers & Rodrigues 2008, Oliveira et al. 2007). Other conservation programs may restrict timber extraction sufficiently to affect timber prices, which could subsequently yield leakage to other forested lands either within (Murray et al. 2004) or beyond the frontiers of the country where the program is implemented (Gan & McCarl 2007, Sohngen et al. 1999). Such spillovers might affect the utilization not just of plantation lands in those spillover locations but also of natural forests (Sohngen et al. 1999). The extent of these effects will depend on the price elasticities of the supply and demand of forest products (Baylis et al. 2013, Gan & McCarl 2007), and the sizes of those effects might vary considerably (Murray et al. 2004).

Leakage involving other policies. Large conservation programs, such as the establishment of PAs, might also affect the plans for public officials to invest in infrastructure, which in turn, might increase or decrease deforestation. Where tourism is important, as it is in Costa Rica (Robalino & Pfaff 2012), tourism infrastructure may develop around PAs. In the United States, there is evidence that housing may grow around PAs (Radeloff et al. 2010). Generally, green spaces may increase the likelihood of development by raising house values (Geoghegan 2002, Irwin 2002, Irwin & Bockstael 2001), although there is also evidence that open space reduces housing development (Lewis et al. 2009).

Facilitating tourism via infrastructure investments, so that more tourists arrive with a positive willingness to pay to see forested land, can increase local incentives to conserve. Private landholders may leave lands in forest in order to benefit from that willingness to pay. For example, there is evidence that, around the same Costa Rican PAs that generated leakage in some locations, there was no leakage near the entrance of those PAs where tourism is concentrated (Robalino et al. 2015a).

If newly facilitated tourism increases labor demand, this increase might offset the decrease in labor demand discussed above. Thus, instead of a falling wage that spurs migration and clearing elsewhere (Robalino 2007), if ecotourism activities are significant, then local wages might increase (Robalino & Villalobos 2015). However, in this case, migration to new jobs in tourism could affect the net impact on wages. If higher wages result, then the costs of agriculture will increase, and deforestation will decrease. This is one form of the class of mechanisms that could result from a carbon tax (see Baylis et al. 2014) that negatively influences the size of other sectors in an economy and therefore reduces emissions.

Large conservation programs might affect the infrastructure for both conservation and development too. For instance, knowing that private migration choices could respond negatively to the creation of a new PA, agencies responsible for transportation to stimulate development could decide to respond to new PAs by reducing their investments related to the creation or maintenance of roads. Herrera (2015) suggests that this happened in the Brazilian Amazon. If private landowners expect these policy responses, and thus expect changes in demands for labor and output, this can clearly generate spillovers from conservation programs.

Finally, both public investment and public enforcement can play a role in spillovers from conservation programs. Especially in developing countries, the enforcement of conservation programs can be lacking. This affects local expectations about such issues as whether a monetary fine might arise from failing to conserve forest. Such expectations provide a channel for spillovers, as any enforcement of a conservation program can lead landowners to believe that other forms of enforcement (in other locations or of other policies) are more likely than previously thought. Within the United States, a nice example of such an effect is provided by Shimshack & Ward (2005). They find evidence that polluters who are not fined react to fines on other actors. Similarly, in the context of the blacklisting of the counties with higher deforestation in the Brazilian Amazon, which includes increased monitoring and costly punishments, Andrade & Chagas (2016) find that deforestation also decreases in counties next to the blacklisted counties and suggest that this is due to enforcement expectations.

In Costa Rica, a role for enforcement within spillovers from conservation programs arose via quite a different mechanism, namely an interaction between the creation of PAs and a regulation that prohibited land-use change near them. In the periods before the latter law had passed, lands next to PAs were not deforested less than lands far away from parks (Robalino et al. 2015a), and in fact, sometimes there was leakage or higher deforestation. However, after the law passed, deforestation next to PAs was significantly lower than in similar land far away from parks (Robalino et al. 2015b). This is consistent with PAs having enforcement spillovers in a sense that, given the law, personnel who monitored the PAs could also monitor forested lands next to PAs.

Channel 3: Learning

Conservation programs can involve practices that a landowner may not have implemented before. Some practices lower deforestation, such as those promoted in certification of logging to reduce damage from timber extraction. Others facilitate forest regrowth, such as multiple approaches to the planting of trees on working lands (see Pagiola et al. 2016 for a discussion of these options). In addition, learning spillovers naturally arise within many other settings outside of forest and land use. Examples include emissions reductions in the US Environmental Protection Agency's 33/50 Program (see Zhou et al. 2017) or choices made by Bangladeshi households to avoid arsenic-contaminated wells (see Balasubramanya et al. 2013).

Before trying a practice—or, if untreated, before seeing others try a practice—private firms or individuals may lack information about the costs and the benefits of the technologies involved. Some practices may yield net benefits even in the absence of any compensation for undertaking them. If so, those practices might continue after the incentives provided by conservation programs end (temporal spillovers). They might also be taken up by actors who are not in the program in question (spatial spillovers) or be applied to treated actors' untreated lands. The latter is a form of slippage, and the new surplus could provide capital to invest in these practices. However, we note that this slippage would have the opposite effect from the slippage discussed above, in that it would help the forest.

The behavioral focus here is the adoption of practices, within which it is easy to imagine such spillovers. Pannell et al. (2006), for instance, discuss three stages within adoptions of new practices: awareness of the opportunity, nontrial evaluation, and trial evaluation. Conservation programs could play a helpful role within each stage. For example, farmers who observe new conservation practices on other farms learn about those options as well as the costs and returns on investment for those who adopt. Even for a single property or landowner, conservation programs could lead to experimentation with new practices on a few parcels. Such experiences often affect the landowner's decisions about the rest of their lands.

Outside of conservation but relevant for the adoption of agricultural technologies that could be forest friendly, there exists a significant body of evidence on how new technologies spread between neighbors in agriculture (e.g., Bandiera & Rasul 2006; Beaman et al. 2015; BenYishay & Mobarak 2015; Busch & Vance 2011; Conley & Udry 2010; Foster & Rosenzweig 1995, 2010; Matuschke & Qaim 2009; Munshi 2004). Similarly, neighbors and social networks clearly can affect the adoption of conservation practices on working lands. Thus, conservation practices by one landowner might well affect the land-use practices of neighboring landowners.

There exists such empirical evidence for settings involving resource extraction, suggesting spillovers from neighbors' behaviors in forests (Robalino & Pfaff 2012), groundwater pumping (Pfeiffer & Lin 2012), and petroleum extraction (Lin 2009, 2013). These spillovers could involve learning about profits, based on others' choices and outcomes, though again, other mechanisms are possible. Yet various mechanisms imply that conservation programs affecting the behavior of one person can also affect others indirectly.

For example, information is generally important for the adoption of conservation practices (Bekele & Drake 2003, Cramb 2006). Furthermore, empirical evidence implies that neighbors might be a channel for such information to flow, given evidence that a key determinant of conservation practices is how likely one's neighbors are to engage in conservation practices. There is evidence of such behavior in conservation tillage (Tessema et al. 2016) and organic dairy farming (Lewis et al. 2011). There is also evidence that the size of the social network affects the likelihood of engaging in natural resource management practices in Ethiopia (Wossen et al. 2013).

That said, learning about productive agricultural technologies might not always be positive for conservation. Farmers also learn about benefits from technologies that could generate deforestation. Busch & Vance (2011) show that, as the fraction of villagers who engage in cattle ranching increases, the higher are the chances of other villagers adopting cattle ranching in the next period. However, they also find that these effects are not linear, and that at some point (64% in their case), the effect of more people from a village involved in cattle ranching decreases its likelihood within the next period.

An important caveat to the claims discussed in this literature is that many empirical claims about learning are based on the inferred effects of neighbors' adoptions. As discussed in Maertens & Barrett (2013), Robalino & Pfaff (2012), and other articles, the identification of neighbors' effects can be challenging. The presence of simultaneous influence across neighbors and unobservable cofounders that affect entire neighborhoods might easily bias such estimates. Moreover, even if the impact of one neighbor's adoption on others was well estimated, there might be other reasons why neighbors affect each other that are not actually about learning. Thus, empirical estimates of spillovers from adoption could easily represent multiple mechanisms.

Finally, we reiterate that the temporal element of spillovers through learning implies that, once incentives within conservation programs are eliminated, farmers might still engage in conservation practices if they have learned that those practices are profitable. There is evidence of this within the aforementioned CRP in the United States (Jacobson 2014). For Colombia, this also arises for some conservation practices within silvopastoral systems that were promoted through incentives based on a suite of potential practices (Pagiola et al. 2016, Zapata et al. 2015). Of course, this will not always hold. In one case, once a set of communities in Mexico withdrew from a particular program of PES, the gains were lost (Le Velly et al. 2015).

Channel 4: Nonpecuniary Motivations

Conservation programs have multiple dimensions, ranging from the provision of information about the actions of neighbors to the location of land-use restrictions and the choice of who is permitted to participate. In some settings, these and other program dimensions, as well as the mere existence of a public program that would restrict private actions, or an external actor who would restrict local actors, could influence a local individual's motivation to carry out the behaviors or practices that are suggested by the program.

In addition, the individuals or firms in question may compare themselves with others. That could yield positive or negative perceptions of programs in terms of fairness or equity. For instance, should a conservation program be judged to be unfair, the nonpecuniary motivation of unfairness could well outweigh other incentives within compliance decisions. Here, we consider situations in which programs appear to have shifted nonpecuniary motivations to engage in conservation.

There is a relatively recent and now large economics literature outside of conservation that highlights the importance of nonmonetary concerns. There is also evidence of such influences within conservation (e.g., Bruner & Reid 2015, Ferraro 2014, Thøgersen & Crompton 2009). For example, empirical evidence supports norm-based theories suggesting that people judge the appropriateness of actions through comparisons to others (Ferraro et al. 2011, Festinger 1954). Judging by comparisons is another reason (in addition to learning) to expect individuals to imitate the choices of others, at least others who are seen as setting community norms. Generally, conservation programs that support norms might generate spillovers that enhance a given program's goals, whereas programs that violate such norms could be subject to retaliatory leakage.

This is relevant for resource extraction decisions by members of a sector or community. Examples include technique choice in mining (Rodriguez 2016) and collective decisions upstream in watersheds (Kaczan et al. 2016, 2017; Pfaff et al. 2017a). Whether a given individual is willing to incur costs to help forests may be influenced by whether those around them are willing to do so; they may even set a norm by doing so. Again, conservation programs inducing behaviors that generate positive imitations can have higher net impacts.

Such norms also can affect decisions to participate in conservation programs (Chen et al. 2009, 2012). Game-theoretic or strategic explanations for such decisions are also possible (existing data and methods often cannot distinguish between multiple possible channels), including when policy units such as states or agencies interact. Nonetheless, norms may play a role not only for individuals but also for local governments (Sauquet et al. 2014).

A linked point about the importance of process within the implementation of conservation programs is suggested by evidence showing that if people are allowed to communicate, they tend to reach agreements about resource extraction with better social outcomes (Cardenas et al. 2000, Ostrom 1990). Furthermore, additional information or guidance also could matter. For example, when individuals were given the opportunity to (noncoercively) talk with a park ranger before they communicated and decided, extraction was further reduced (Moreno-Sánchez & Maldonado 2010). Such results arise even though individuals gain monetarily from extracting more than is socially optimal. Thus, these results provide evidence about the importance of social norms in resource contexts.

However, as in other areas of life (e.g., Gneezy & Rustichini 2000), it appears that local norms can be crowded out by even the mere creation of a government program to influence local behaviors and outcomes. Such crowding out represents one form of spillovers. For instance, in a framed field experiment, Cardenas et al. (2000) find that a new fine for behaviors that damage the forest leads to more of those behaviors than were chosen when individuals were subject only to their sense of local norms. Along the same lines, the mere existence (or sometimes existence then removal) of monetary incentives such as payments might crowd out other motivations to conserve. If such effects exist and continue, negative spillover effects will occur over time (Kaczan et al. 2016, Kits et al. 2014).

Across a variety of conservation settings, information about some other peoples' behavior can potentially guide one's own behavior, even when that information clearly has no pecuniary effects. Agencies seem to believe this, as evidenced by the existence of programs designed to provide such information about neighbors to influence, for example, the conservation of electricity (Allcott 2011, Ayres et al. 2012, Costa & Kahn 2013) or water (Bernedo et al. 2014, Ferraro & Miranda 2013, Ferraro & Price 2013, Goldstein et al. 2008, Jaime & Carlsson 2014). Overall, studies show that if peer information is provided, this alone can lead individuals to change their levels of consumption. Thus, sharing information about the behaviors of others could lead to lower or higher equilibria in overall conservation efforts, which demonstrates the potential for large spillovers if a program shifts equilibria.

Furthermore, the relevant consideration of others includes the interactions through norms, whether to trust other individuals, and how to weight others' welfare if other regarding preferences play a significant role in behavior. That is relevant for conservation, in which decisions may well affect the happiness of other people. In water allocation, for instance, trust in the sharing of surplus from efficient resource allocation can be a key determinant of outcomes when multiple parties are involved in resource use (Pfaff & Vélez 2012, Pfaff et al. 2017b).

Emotion-based interactions among individuals imply spillovers via their influences on each other. These can even affect optimal institutions, in light of efficiency and equity concerns (Pfaff et al. 2017b). For example, conservation programs that help support cooperative behavior in initial periods can generate trust and even changes in how individuals treat each other in the future (Pfaff et al. 2015c). As in learning, there can be temporal and spatial spillovers from early decisions. This implies the potential for significant influences on total net impacts from early program design.

Finally, another important possible mechanism for program spillovers is the consideration of fairness. Many settings exist in which a conservation program must be targeted; in other words, it is offered to only a subset of possible participants. A common reason for this is simply that resources are limited, such as for transfers within PES. Any such targeting implies that some people are excluded. If they feel that the exclusion was unfair, then they could consequently shift behavior, despite no change in pecuniary incentives. Alpízar et al. (2015, 2017) find such an effect for farmers making proforest donations in Costa Rica, specifically finding spillovers as a function of why people were excluded (given that all individuals know the randomized exclusion rules). Rules that are seen as fair, such as the random selection of beneficiaries, do not generate negative effects. However, selecting those individuals who exhibited low prosocial behavior and who may need incentives to change their behavior to aid society yields a negative response from the more prosocial individuals.

Channel 5: Ecological-Physical Links

Finally, although we devote less space to these channels because they do not involve human behavior, we also note the possible spillover mechanisms inherent within ecological or physical processes. Similar to prices (i.e., perhaps analogous to economic systems), these function as transmission mechanisms that link locations and time periods. For instance, when species richness increases in one site in one period (e.g., if an area is successfully targeted by conservation programs), the biodiversity in nontargeted forest corridors can benefit as well (Brudvig et al. 2009). Similarly, within fisheries, ecosystem interactions can propagate the effects of conservation programs to neighboring areas (e.g., Sanchirico 2004; Sanchirico & Wilen 1999, 2001). Once again, those spillovers could well outlast the programs (temporal spillovers).

In addition to such ecologically based spillover effects due to such factors as shifts in species' migration and reproduction behaviors, purely physical processes could also spread the impacts of conservation programs. For example, underground extraction of both oil and groundwater is subject to the laws of pressure. These laws imply that extraction in one location shifts marginal costs of extraction and thus shifts extraction elsewhere (Foster & Sekhri 2008; Foster & Rosenzweig 2005; Lin 2009, 2013; Pfeiffer & Lin 2012).

Within any optimal program design, such ecological and physical links need to be considered (Costello & Polasky 2008, Polasky et al. 2008). Not only are outcomes and thus welfare affected, but those links may also lead to further behavioral response and thus additional program spillovers.

DISCUSSION

In reviewing the literature relevant to conservation spillovers, with a focus on empirical evidence, we find that the amount of evidence for spillovers is growing. That is a positive sign of attention to these issues. We have emphasized that many different channels can generate spillovers, perhaps helping to explain the burgeoning set of analyses of different phenomena. We have focused on five channels: (Channel 1) input reallocation; (Channel 2) market prices; (Channel 3) learning; (Channel 4) nonpecuniary motivations; and (Channel 5) ecological-physical links. Each channel was considered in detail.

Below, we provide some additional discussion and commentary. We emphasize that spillovers are heterogeneous in their existence and even in their direction. Sometimes, for example, they support program goals. When spillovers run counter to program goals, we consider how to avoid or minimize them. Whatever their direction, we discuss challenges in the empirical identification of spillovers' magnitudes as well as in the clear communication of findings. Finally, we stress the need for more research that carefully reflects the variety of contexts within which spillovers may occur.

Heterogeneity

The empirical literature makes clear that spillovers from conservation programs may or may not occur, depending on the setting. Moreover, if they do exist, they will be heterogeneous in their magnitudes and even in their direction, supporting or running counter to a program's goals.

One perhaps obvious but still important point that seems common across all of the possible channels is that, to generate significant spillovers, conservation programs must have significant initial impacts within their boundaries. That is worth noting, primarily because it is not at all clear that past programs have had large impacts (e.g., see Pfaff et al. 2013 for one review of many kinds of programs). Thus, we may not expect spillovers.

There are categories of channels for which positive spillovers (supporting program goals) may be expected and thus also celebrated and further encouraged. Learning about the profitability of new practices promoted by conservation programs certainly is one such category, which likely explains significant hopeful attention to the idea of climate-smart agriculture practices that by assumption generate productivity gains alongside lower environmental impacts. Another category involves strategic complementarity of conservation actions. An example of this would be when a private or public landowner chooses to conserve forest that in turn increases the benefits and returns to the forest for the neighboring landowners (Robalino & Pfaff 2012).

Other categories can mix positive and negative spillovers depending on distance to a program; for example, they can yield different directions for local spillovers versus spillovers farther away. Establishing a PA can lower labor demand and also may signal that the region containing the new PA will be considered green by the relevant state, which reduces expectations of future local investments in development. Should that lead to out-migration (Herrera 2015), then deforestation might fall near the PA (i.e., positive forest spillovers). Yet because deforestation is likely to increase elsewhere, a view of net welfare impact from a conservation intervention requires a comparison of these effects. For instance, if the PA was sited near favored species, the local gains might be of more importance.

Avoidance

The ways in which negative spillovers (countering program goals) may be avoided vary by channel. If the nonpecuniary motivations channel is relevant, then it seems possible that designing programs around fairness and communication of rationales might ameliorate negative spillover effects. Alternatively, for cases when the generation of the spillovers would involve other public actions, such as investment or enforcement, those other actions could be adjusted or coordinated with the program in question.

For other channels, limiting negative spillovers could involve monitoring and enforcement. If concerned about input slippage, for example, in considering the mandate in the Brazilian Amazon forest code that some property remains in forest, then avoiding within-property spillovers requires detailed monitoring of private properties. That might be possible given improved remote sensing. Contrasting this case to the aforementioned idea of comparing different deforestation impacts for a net effect, we note recent proposals to allow the trading of forest code requirements so that total output rises. This is because landowners who can produce more when they are allowed to further deforest can buy clearing permits from those who can produce less (thus, the trading policy's focus is the distinct impacts on economic output). That intentionally trades local changes for distant forest changes, assuming that all forest is equal.

We suppose that monitoring and enforcement would be harder for physical and ecological processes. For instance, netting might restrict the movements of fish and birds to reduce spillover effects, were that desirable, yet the challenges could be significant. Limiting market-based leakage might also be very difficult, as it is hard to limit price increases given reductions in supply due to conservation. Nonetheless, as within optimal tax policy, conservation policies could consider the relevant elasticities of supply and demand that would affect the extent of market-based leakages.

Identification

Because policy recommendations often require an understanding of channels, we emphasize that many empirical estimates of spillovers cannot distinguish channels. That is often a difficult challenge and may be impossible depending on available data. For protected areas, for instance, although there is some evidence about impacts on wages (Robalino & Villalobos 2015), to understand exactly how and why farmers move inputs between parcels would require far more data than typically have been available, particularly within developing countries. With better data sets being compiled over time, we believe that channels will become better identified, at least if future research prioritizes separating the influences of varied channels.

Even if such data were available, however, identifying the magnitude of spillovers is complex. One challenge is that the places that might be affected (or might have been affected) by spillovers are not always easy to know. Not knowing where to collect data is a real challenge, and looking in the wrong places often means overlooking existing effects when they exist elsewhere. Thinking about the likely mechanisms in advance can, at least in principle, offer some guidance.

Another issue in identifying spillovers is that spatial influence can go in both directions; the use of each neighbor's parcel affects incentives for each other neighbor. It can be impossible to empirically separate effects in one direction (Manski 1993). Another challenge for identification is spatially correlated unobservable factors that generate spatial associations that can resemble spillovers even if there are none (or, if spillovers do actually exist, such factors can confound interpretation). Sometimes an instrumental variable might suggest itself, such as the use of the neighboring slopes discussed in Robalino & Pfaff (2012).

As a side note, the existence of local spillovers presents a clear challenge for another frequently attempted identification strategy, which might be called spatial regression discontinuity: Treated lands are compared to lands that are adjacent but on the other side of a treatment boundary. This could help to control for spatially varying unobservables. Yet with spillovers, it will not work. However, another sign of broader recognition of the empirical and policy importance of spillovers is that now some research is being designed to avoid contamination of controls by spillovers. In the case of learning, research measures spillovers as part of the total impact (across contexts, including energy and voting; see Baird et al. 2014, Banerjee et al. 2014, Carranza & Meeks 2016, and Giné & Mansuri 2011).

Communication

Communication about spillovers for the purpose of ranking policy options may benefit from more uniform reporting of the empirical findings from spillover studies. Findings are currently reported in various ways, such as the percentage of protected land, percentage of the impact of conservation policies, or in other (understandable) forms. Because one key issue is gaining a sense of how much the spillover changes a program's net impact, there might be a gain if all studies included the percentage of the within-boundary impact.

Context

The variation in different studies' ways of reporting on spillovers may reflect a fundamental issue that deserves mention on its own. Namely, spillovers occur in a wide range of settings, and their existence, magnitudes, and even directions can vary across the settings. Thus, even if empirical identification was perfect for the spillovers from one specific policy at one specific place and time, those results for the magnitude and even direction of spillovers might not apply for other important settings.

Along these lines, we end this review by highlighting external validity using one PA example. Robalino et al. (2015a) guided their breakdown of the average local deforestation spillover from Costa Rican PAs (Andam et al. 2008) on the basis of this consideration of channels; the prior approach was correct but blended heterogeneous effects. Although that breakdown illuminated meaningful previously hidden effects for the Costa Rican setting, those results did not translate to local PA spillovers within the Brazilian Amazon, even though studies have found similar results in those countries for the within-boundary impacts of PAs. The reason is that the relevant economic and political dynamics on that developing frontier are very different from those in Costa Rica—so much so that even the direction is different for the average local spillover from Brazilian Amazon

PAs (Herrera 2015). Future research could include a significant focus on describing the relevant features of any setting, as the specific setting influences not only which channels arise but also which ones dominate the net spillover impacts.

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LITERATURE CITED

Alix-Garcia JM, Shapiro EN, Sims KR. 2012. Forest conservation and slippage: evidence from Mexico's national payments for ecosystem services program. *Land Econ*. 88(4):613–38

Allcott H. 2011. Social norms and energy conservation. *J. Public Econ.* 95(9):1082–95

Alpízar F, Nordén A, Pfaff A, Robalino J. 2015. Unintended effects of targeting an environmental rebate. *Environ. Resour. Econ.* 67(1):181–202

Alpízar F, Nordén A, Pfaff A, Robalino J. 2017. Spillovers from targeting of incentives: exploring responses to being excluded. *J. Econ. Psychol.* 59:87–98

Andam KS, Ferraro PJ, Pfaff A, Sanchez-Azofeifa GA, Robalino JA. 2008. Measuring the effectiveness of protected area networks in reducing deforestation. *PNAS* 105(42):16089–94

Andrade L, Chagas ALS. 2016. *Spillover effects of blacklisting policy in the Brazilian Amazon*. Work. Pap. No. 2016_32, Univ. São Paulo, Braz.

Armsworth PR, Daily GC, Kareiva P, Sanchirico JN. 2006. Land market feedbacks can undermine biodiversity conservation. *PNAS* 103(14):5403–8

Arriagada RA, Ferraro PJ, Sills EO, Pattanayak SK, Cordero-Sancho S. 2012. Do payments for environmental services affect forest cover? A farm-level evaluation from Costa Rica. *Land Econ*. 88(2):382–99

Ayres I, Raseman S, Shih A. 2012. Evidence from two large field experiments that peer comparison feedback can reduce residential energy usage. *J. Law Econ. Organ.* 29(5):992–1022

Baird S, Bohren A, McIntosh C, Ozler B. 2014. *Designing experiments to measure spillover effects*. PIER Work. Pap. 14-032

Balasubramanya S, Pfaff A, Bennear L, Tarozzi A, Ahmed KM, et al. 2013. Evolution of households' responses to the groundwater arsenic crisis in Bangladesh: information on environmental health risks can have increasing behavioral impact over time. *Environ. Dev. Econ.* 19(5):631–47

Bandiera O, Rasul I. 2006. Social networks and technology adoption in northern Mozambique. *Econ. J.* 116(514):869–902

Banerjee A, Chattopadhyay R, Duflo E, Keniston D, Singh N. 2014. *Improving police performance in Rajasthan, India: experimental evidence on incentives, managerial autonomy and training*. NBER Work. Pap. 17912

Baylis K, Fullerton D, Karney DH. 2014. Negative leakage. *J. Assoc. Environ. Resour. Econ.* 1(1/2):51–73

Baylis K, Fullerton D, Shah P. 2013. *What drives forest leakage?* Work. Pap., Dep. Agric. Consum. Econ., Univ. Ill., Urbana-Champaign

Beaman L, BenYishay A, Magruder J, Mobarak AM. 2015. *Can network theory based targeting increase technology adoption?* Work. Pap., Northwestern Univ., Chicago, Ill.

Bekele W, Drake L. 2003. Soil and water conservation decision behavior of subsistence farmers in the Eastern Highlands of Ethiopia: a case study of the Hunde-Lafto area. *Ecol. Econ.* 46(3):437–51

BenYishay A, Mobarak AM. 2015. *Social learning and incentives for experimentation and communication*. Work. Pap., Yale Univ., New Haven, Conn.

Bernedo M, Ferraro PJ, Price M. 2014. The persistent impacts of norm-based messaging and their implications for water conservation. *J. Consum. Policy* 37(3):437–52

Brudvig LA, Damschen EI, Tewksbury JJ, Haddad NM, Levey DJ. 2009. Landscape connectivity promotes plant biodiversity spillover into non-target habitats. *PNAS* 106(23):9328–32

Bruner A, Reid J. 2015. *Behavioral economics and payments for ecosystem services: finally some free lunches*. Disc. Pap. 13, Conserv. Strat. Fund, Washington, DC

Busch CB, Vance C. 2011. The diffusion of cattle ranching and deforestation: prospects for a hollow frontier in Mexico's Yucatán. *Land Econ*. 87(4):682–98

Cardenas JC, Stranlund J, Willis C. 2000. Local environmental control and institutional crowding-out. *World Dev*. 28(10):1719–33

Carranza E, Meeks R. 2016. *Shedding light: understanding energy efficiency and electricity reliability*. Policy Res. Work. Pap., World Bank, Washington, DC

Chen X, Lupi F, An L, Sheely R, Vina A, Liu J. 2012. Agent-based modeling of the effects of social norms on enrollment in payments for ecosystem services. *Ecol. Model.* 229:16–24

Chen X, Lupi F, He G, Liu J. 2009. Linking social norms to efficient conservation investment in payments for ecosystem services. *PNAS* 106(28):11812–17

Conley TG, Udry CR. 2010. Learning about a new technology: pineapple in Ghana. *Am. Econ. Rev.* 100(1):35–69

Costa DL, Kahn ME. 2013. Energy conservation "nudges" and environmentalist ideology: evidence from a randomized residential electricity field experiment. *J. Eur. Econ. Assoc.* 11(3):680–702

Costello C, Polasky S. 2008. Optimal harvesting of stochastic spatial resources. *J. Environ. Econ. Manag.* 56(1):1–18

Cramb RA. 2006. The role of social capital in the promotion of conservation farming: the case of 'landcare' in the Southern Philippines. *Land Degrad. Dev.* 17(1):23–30

De Sá SA, Palmer C, Di Falco S. 2013. Dynamics of indirect land-use change: empirical evidence from Brazil. *J. Environ. Econ. Manag.* 65(3):377–93

Desbureaux S, Kere EN, Motel PC. 2016. *Impact evaluation in a landscape: protected natural forests, anthropized forested lands and deforestation leakages in Madagascar's rainforests*. Work. Pap. 238, Afr. Dev. Bank Group, Abidjan, Côte-d'Iv.

Ewers RM, Rodrigues AS. 2008. Estimates of reserve effectiveness are confounded by leakage. *Trends Ecol. Evol.* 23(3):113–16

Ferraro P. 2014. The road to sustainability: more nudging, less shoving. *Snap.is Magazine*, Jan. 27. **http://snappartnership.net/magazine/paul-ferraro-sustainability-nudges-conservation/**

Ferraro P, Hanauer MM, Miteva DA, Canavire-Bacarezza GJ, Pattanayak SK, Sims KRE. 2013. More strictly protected areas are not necessarily more protective: evidence from Bolivia, Costa Rica, Indonesia and Thailand. *Environ. Res. Lett.* 8:025011

Ferraro PJ, Miranda JJ. 2013. Heterogeneous treatment effects and mechanisms in information-based environmental policies: evidence from a large-scale field experiment. *Resour. Energy Econ.* 35(3):356–79

Ferraro PJ, Miranda JJ, Price MK. 2011. The persistence of treatment effects with norm-based policy instruments: evidence from a randomized environmental policy experiment. *Am. Econ. Rev.* 101(3):318–22

Ferraro PJ, Pattanayak SK. 2006. Money for nothing? A call for empirical evaluation of biodiversity conservation investments. *PLOS Biol*. 4(4):e105

Ferraro PJ, Price MK. 2013. Using nonpecuniary strategies to influence behavior: evidence from a large-scale field experiment. *Rev. Econ. Stat.* 95(1):64–73

Festinger L. 1954. A theory of social comparison processes. *Hum. Relat.* 7(2):117–40

Fleming DA. 2010. *Slippage effects of the conservation reserve program: new evidence from satellite imagery*. Presented at Annu. Meet. Agric. Appl. Econ., Denver, Colo.

Foster AD, Rosenzweig MR. 1995. Learning by doing and learning from others: human capital and technical change in agriculture. *J. Polit. Econ.* 103(6):1176–209

Foster AD, Rosenzweig MR. 2005. *Inequality and sustainability of agricultural growth: groundwater and green revolution in rural India*. Work. Pap., Brown Univ., Providence, RI

Foster AD, Rosenzweig MR. 2010. Microeconomics of technology adoption. *Annu. Rev. Econ.* 2:395–424

Foster AD, Sekhri S. 2008. *Can expansion of markets for groundwater decelerate the depletion of groundwater resource in rural India?* Work. Pap., Brown Univ., Providence, RI

Gan J, McCarl BA. 2007. Measuring transnational leakage of forest conservation. *Ecol. Econ.* 64(2):423–32

Geoghegan J. 2002. The value of open spaces in residential land use. *Land Use Policy* 19(1):91–98

Giné X, Mansuri G. 2011. *Together we will: experimental evidence on female voting behavior in Pakistan*. Policy Res. Work. Pap. 5692, World Bank, Washington, DC

Gneezy U, Rustichini A. 2000. Fine is a price. *J. Legal Stud.* 29:1–17

Goldstein NJ, Cialdini RB, Griskevicius V. 2008. A room with a viewpoint: using social norms to motivate environmental conservation in hotels. *J. Consum. Res.* 35(3):472–82

Herrera D. 2015. *Protected areas' deforestation spillovers and two critical underlying mechanisms: an empirical exploration for the Brazilian Amazon*. PhD Thesis, Duke Univ., Durham, NC

Irwin EG. 2002. The effects of open space on residential property values. *Land Econ*. 78(4):465–80

Irwin EG, Bockstael NE. 2001. The problem of identifying land use spillovers: measuring the effects of open space on residential property values. *Am. J. Agric. Econ.* 83(3):698–704

Jacobson S. 2014. Temporal spillovers in land conservation. *J. Econ. Behav. Organ.* 107:366–79

Jaime M, Carlsson F. 2014. *Social norms and information diffusion in water-saving programs: evidence from a randomized field experiment in Colombia*. Work. Pap., Dep. Econ., Univ. Gothenburg, Swed.

Kaczan DJ, Pfaff A, Rodriguez L, Shapiro-Garza E. 2017. *Increasing the impact of collective incentives in payments for ecosystem services*. Work. Pap., Duke Univ., Durham, NC

Kaczan DJ, Swallow BM, Adamowicz WL. 2016. Forest conservation policy and motivational crowding: experimental evidence from Tanzania. *Ecol. Econ.* **http://dx.doi.org./10.1016/j.ecolecon.2016.07.002**

Kits GJ, Adamowicz WL, Boxall PC. 2014. Do conservation auctions crowd out voluntary environmentally friendly activities? *Ecol. Econ.* 105:118–23

Lambin EF, Meyfroidt P. 2011. Global land use change, economic globalization, and the looming land scarcity. *PNAS* 108(9):3465–72

Le Velly G, Sauquet A, Cortina-Villar S. 2015. *PES impact and leakages over several cohorts: the case of PSA-H in Yucatan, Mexico*. Work. Pap. 29, CERDI, Clermont-Ferrand, Fr.

Lewis DJ, Barham BL, Robinson B. 2011. Are there spatial spillovers in the adoption of clean technology? The case of organic dairy farming. *Land Econ*. 87(2):250–67

Lewis DJ, Provencher B, Butsic V. 2009. The dynamic effects of open-space conservation policies on residential development density. *J. Environ. Econ. Manag.* 57(3):239–52

Lichtenberg E, Smith-Ramirez R. 2011. Slippage in conservation cost sharing. *Am. J. Agric. Econ.* 93(1):113–29

Lim FKS, Carrasco LR, McHardy J, Edwards DP. 2017. Perverse market outcomes from biodiversity conservation interventions. *Conserv. Lett.* In press. **https://doi.org/10.1111/conl.12332**

Lin CYC. 2009. Estimating strategic interactions in petroleum exploration. *Energy Econ*. 31(4):586–94

Lin CYC. 2013. Strategic decision-making with information and extraction externalities: a structural model of the multistage investment timing game in offshore petroleum production. *Rev. Econ. Stat.* 95(5):1601–21

Maertens A, Barrett CB. 2013. Measuring social networks' effects on agricultural technology adoption. *Am. J. Agric. Econ.* 95(2):353–59

Manski CF. 1993. Identification of endogenous social effects: the reflection problem. *Rev. Econ. Stud.* 60(3):531–42

Matuschke I, Qaim M. 2009. The impact of social networks on hybrid seed adoption in India. *Agric. Econ.* 40(5):493–505

Meyfroidt P, Lambin EF, Hertel T, Erb K-H. 2013. Globalization of land use: displacement and distant drivers, 2013. *Curr. Opin. Environ. Sustain*. 5(5):438–44

Moreno-Sánchez R, Maldonado JH. 2010. Evaluating the role of co-management in improving governance of marine protected areas: an experimental approach in the Colombian Caribbean. *Ecol. Econ.* 69(12):2557–67

Mullan K, Kontoleon A. 2009. *Participation in Payments for Ecosystem Services programmes in developing countries: the Chinese Sloping Land Conversion Programme*. Work. Pap., Dep. For. Environ. Resour., NC State Univ., Raleigh

Munshi K. 2004. Social learning in a heterogeneous population: technology diffusion in the Indian Green Revolution. *J. Dev. Econ.* 73(1):185–213

Murray BC, McCarl BA, Lee HC. 2004. Estimating leakage from forest carbon sequestration programs. *Land Econ*. 80(1):109–24

Oliveira PJ, Asner GP, Knapp DE, Almeyda A, Galván-Gildemeister R, et al. 2007. Land-use allocation protects the Peruvian Amazon. *Science* 317(5842):1233–36

Ostrom E. 1990. *Governing the Commons. The Evolution of Institutions of Collective Action*. Cambridge, UK: Cambridge Univ. Press

Pagiola S, Honey-Rosés J, Freire-González J. 2016. Evaluation of the permanence of land use change induced by payments for environmental services in Quindío, Colombia. *PLOS ONE* 11(3):e0147829

Pannell DJ, Marshall GR, Barr N, Curtis A, Vanclay F, Wilkinson R. 2006. Understanding and promoting adoption of conservation practices by rural landholders. *Anim. Prod. Sci.* 46(11):1407–24

Pfaff A, Amacher GS, Sills EO. 2013. Realistic REDD: improving the forest impacts of domestic policies in different settings. *Rev. Environ. Econ. Policy* 7(1):114–35

Pfaff A, Robalino J. 2012. Protecting forests, biodiversity, and the climate: predicting policy impact to improve policy choice. *Oxf. Rev. Econ. Policy* 28(1):164–79

Pfaff A, Robalino J, Herrera D, Sandoval C. 2015a. Protected areas' impacts on Brazilian Amazon deforestation: examining conservation–development interactions to inform planning. *PLOS ONE* 10(7):e0129460

Pfaff A, Robalino J, Sanchez-Azofeifa GA, Andam KS, Ferraro PJ. 2009. Park location affects forest protection: land characteristics cause differences in park impacts across Costa Rica. *BE J. Econ. Anal. Policy* 9(2):5. **https://doi.org/10.2202/1935-1682.1990**

Pfaff A, Robalino J, Sandoval C, Herrera D. 2015b. Protected area types, strategies and impacts in Brazil's Amazon: public protected area strategies do not yield a consistent ranking of protected area types by impact. *Phil. Trans. R. Soc. B* 370(1681):20140273

Pfaff A, Rodriguez LA, Shapiro-Garza E. 2017a. *Creating local PES institutions and increasing impacts of PES in Mexico: a real-time watershed-level framed field experiment on coordination and conditionality*. Work. Pap., Duke Univ., Durham, NC

Pfaff A, Vélez MA. 2012. Efficiency and equity in negotiated resource transfers: contributions and limitations of trust with limited contracts. *Ecol. Econ.* 74:55–63

Pfaff A, Vélez MA, Broad K, Hamoudi A, Taddei R. 2017b. *Contracts versus trust: equity and efficiency within resource allocation*. Work. Pap., Duke Univ., Durham, NC

Pfaff A, Vélez MA, Ramos PA, Molina A. 2015c. Framed field experiment on resource scarcity & extraction: path-dependent generosity within sequential water appropriation. *Ecol. Econ.* 120:416–29

Pfeiffer L, Lin CYC. 2012. Groundwater pumping and spatial externalities in agriculture. *J. Environ. Econ. Manag.* 64(1):16–30

Polasky S, Nelson E, Camm J, Csuti B, Fackler P, et al. 2008. Where to put things? Spatial land management to sustain biodiversity and economic returns. *Biol. Conserv.* 141(6):1505–24

Radeloff VC, Stewart SI, Hawbaker TJ, Gimmi U, Pidgeon AM, et al. 2010. Housing growth in and near United States protected areas limits their conservation value. *PNAS* 107(2):940–45

Robalino J, Pfaff A. 2013. Ecopayments and deforestation in Costa Rica: a nationwide analysis of PSA's initial years. *Land Econ*. 89(3):432–48

Robalino J, Pfaff A, Villalobos L. 2015a. *Deforestation spillovers from Costa Rican protected areas*. Work. Pap. 201502, Univ. Costa Rica, San José

Robalino J, Sandoval C, Barton DN, Chacon A, Pfaff A. 2015b. Evaluating interactions of forest conservation policies on avoided deforestation. *PLOS ONE* 10(4):0124910

Robalino J, Villalobos L. 2015. Protected areas and economic welfare: an impact evaluation of national parks on local workers' wages in Costa Rica. *Environ. Dev. Econ.* 20(3):283–310

Robalino JA. 2007. Land conservation policies and income distribution: Who bears the burden of our environmental efforts? *Environ. Dev. Econ.* 12(04):521–33

Robalino JA, Pfaff A. 2012. Contagious development: neighbor interactions in deforestation. *J. Dev. Econ.* 97(2):427–36

Roberts MJ, Bucholtz S. 2005. Slippage in the conservation reserve program or spurious correlation? A comment. *Am. J. Agric. Econ.* 87(1):244–50

Roberts MJ, Bucholz S. 2006. Slippage in the conservation reserve program or spurious correlation? A rejoinder. *Am. J. Agric. Econ.* 88(2):512–14

Rodriguez L. 2016. *On the regulation of small actors: three experimental essays about policies based on voluntary compliance and decentralized monitoring*. PhD Thesis, Duke Univ., Durham, NC

Sanchirico JN. 2004. Designing a cost-effective marine reserve network: a bioeconomic metapopulation analysis. *Mar. Resour. Econ.* 19(1):41–65

Sanchirico JN, Wilen JE. 1999. Bioeconomics of spatial exploitation in a patchy environment. *J. Environ. Econ. Manag.* 37(2):129–50

Sanchirico JN, Wilen JE. 2001. A bioeconomic model of marine reserve creation. *J. Environ. Econ. Manag.* 42(3):257–76

Sauquet A, Marchand S, Féres JG. 2014. Protected areas, local governments, and strategic interactions: the case of the ICMS-Ecológico in the Brazilian state of Paraná. *Ecol. Econ.* 107:249–58

Shimshack JP, Ward MB. 2005. Regulator reputation, enforcement, and environmental compliance. *J. Environ. Econ. Manag.* 50(3):519–40

Sims KR. 2010. Conservation and development: evidence from Thai protected areas. *J. Environ. Econ. Manag.* 60(2):94–114

Sohngen B, Mendelsohn R, Sedjo R. 1999. Forest management, conservation, and global timber markets. *Am. J. Agric. Econ.* 81(1):1–13

Tessema YM, Asafu-Adjaye J, Kassie M, Mallawaarachchi T. 2016. Do neighbours matter in technology adoption? The case of conservation tillage in northwest Ethiopia. *Afr. J. Agric. Resour. Econ.* 11(3):211–25

Thøgersen J, Crompton T. 2009. Simple and painless? The limitations of spillover in environmental campaigning. *J. Consum. Policy* 32(2):141–63

Uchida E, Xu J, Xu Z, Rozelle S. 2007. Are the poor benefiting from China's land conservation program? *Environ. Dev. Econ.* 12:593–620

Uchida S. 2014. *Indirect land use effects of conservation: disaggregate slippage in the US conservation reserve program*. Work. Pap. 14–05, Dep. Agric. Resour. Econ., Univ. Md.

Wossen T, Berger T, Mequaninte T, Alamirew B. 2013. Social network effects on the adoption of sustainable natural resource management practices in Ethiopia. *Int. J. Sustain. Dev. World Ecol.* 20(6):477–83

Wu J. 2000. Slippage effects of the conservation reserve program. *Am. J. Agric. Econ.* 82(4):979–92

Zapata C, Robalino J, Solarte A. 2015. Influencia del Pago por Servicios Ambientales y otras variables biofísicas y socioeconómicas en la adopción de sistemas silvopastoriles a nivel de finca. *Livest. Res. Rural Dev.* 27(4):63

Zhou R, Segerson K, Bi X. 2017. *Evaluating voluntary programs with information spillovers*. Work. Pap., Univ. Conn., Storrs