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Payments for Ecosystem Services and Wealth Distribution

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ABSTRACT

Payment for ecosystem services (PES) has come to be regarded as a promising market-based policy instrument to internalize environmental externalities. The potential of PES is linked to the relationship between the willingness to pay (WTP) of ecosystem service buyers and the willingness to accept (WTA) of ecosystem service providers. This study uses an economic model to analyze factors that influence aggregate WTP and WTA in a PES scheme. We demonstrate that wealth disparity between ecosystem services buyers and providers can increase transactions. Furthermore, when wealth disparity exists between the buyers and sellers, the wealthier population would contribute more into the program and the poorer population would benefit more from it. Under these conditions, PES can be socially progressive and mitigate preexisting economic inequality. In this sense, the economic model provides justification for integration of PES and poverty alleviation programs. Results of our study indicate that PES is not a universally applicable conservation tool, and there is a need for a more targeted approach to the design and application of PES.

1. Introduction

Payment for ecosystem services (PES) is defined as a voluntary transaction of well-defined ecosystem service between providers and beneficiaries (Wunder et al., 2008). The classic example is upstream farmers receiving payments to maintain trees on the landscape in order to conserve downstream communities' drinking water supply and to protect them from flood risk. In a broader sense, some government-financed payment schemes, in which the government makes payments on behalf of beneficiaries to private landowners in order to encourage environmentally friendly land management practices, can also be understood through reference to PES (Muradian et al., 2010; Vatn, 2010). Through providing economic incentives, PES aligns individuals' interests with environmental and social wellbeing of the society. As a market-based policy instrument, PES is also assumed to be more flexible and cost-effective than command-and-control approaches in addressing complex environmental challenges, such as non-point source pollution, biodiversity loss, and greenhouse gas emissions (Daily and Matson, 2008; Goldman et al., 2008). It gives individuals freedom to choose strategies that fit their specific situation, thereby better reflecting the heterogeneity of environmental issues compared to command-and-control approaches (Jack et al., 2008; Vatn, 2010).

Besides environmental management aims, many PES programs also have social targets, most importantly poverty alleviation. Studies suggest that the rural poor are more likely to live on marginal lands that are prone to erosion and degradation (Pagiola et al., 2005; Engel et al., 2008; Milder et al., 2010), and poverty is also a major driver of natural resource exploitation that threatens flows of many types of ecosystem services (Bulte et al., 2008). Thus by paying low-income people to adopt environmentally friendly practices, PES can advance both environmental conservation and poverty alleviation goals. There are both theoretical and empirical studies that support pro-poor PES. For instance, Zilberman and colleagues use an economic model to demonstrate that the poor are more likely to benefit from PES programs if the revenues from ecosystem services and agricultural activities are negatively correlated (Zilberman et al., 2008). Grieg-Gran et al. (2005) reviewed multiple PES programs in Latin American, and found that poor people that participated in PES programs usually benefitted from significant increases in both cash income and social capital. Other empirical studies indicated that even though in some cases PES programs are not intended for poverty reduction, there can be important synergies if the contexts are favorable. Particularly, the poor are more likely to become better off if participation is voluntary (Pagiola et al., 2005; Milder et al., 2010). Realization of such synergies is challenging, of course. Studies highlight the difficulties of making PES socially inclusive and the potential of exacerbating poverty in some cases. For instance, some PES programs have demanding application procedures or require

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substantial initial investments that make it difficult for the poor to participate (Landell-Mills and Porras, 2002). Further, by increasing the value of land identified as the source of valuable services, PES can catalyze investments and so-called "green grabbing" that limit the poor's access to the land on which they depend (Kerr, 2002; Fairhead et al., 2012).

Despite the significant investments in development of PES over the past two decades, such projects encounter substantial obstacles. It is, therefore, important to reflect on the gap between the promise and reality of PES, and to identify the major barriers to the success of PES. Here we identify and briefly review five major constraints. Firstly, the monetary value of ecosystem services provided by an individual land manager is generally very small, and correspondingly the willingness to pay (WTP) for these services is usually very low. The WTP in a PES program is the exchange value, which is largely determined by direct services from ecosystems, such as water purification, soil erosion mitigation, or carbon sequestration. In aggregate, the values of these services to human society are substantial. But at the level of specific parcels of land, the values of these services from a farm field or forest patch are usually low compared to the costs to provide these services. The Kyoto Protocol's Clean Development Mechanism (CDM) offers a useful example, as smallholders have been largely excluded from the carbon sequestration market because the value of the emissions offset they could provide individually is relatively low while the costs to meet the CDM requirements (e.g., analysis, documentation, and monitoring) are high (Henman et al., 2008).

Secondly, from the ecosystem services providers' perspective, their willingness to accept (WTA) is based on the costs of provision, rather than the value of ecosystem services. Some PES-like programs require participants to take land out of production and leave it idle, such as the Conservation Reserve Program in the United States (Cain and Lovejoy, 2004; Flinchbaugh and Knutson, 2004), or require affirmative actions, such as the Slope Land Conversion Program (SLCP) in China that requires reforestation (Bennett, 2008). These requirements can represent significant expenditures and/or opportunity costs to the producers, thus the WTA of the participants could be very high. Furthermore, in some cases, the provision of ecosystem services means giving up certain social, cultural, or traditional identities rather than the service itself. One example is the eco-compensation program in Qinghai, China, where the government pays traditional nomadic herders to reduce herd sizes or to completely quit pastoralism in order to protect degraded grassland. Because nomadic pastoralism has cultural significance to most people within this ethnic population and because employment options in resettlement villages are unclear, their WTA is understandably extremely high, if they can be convinced to participate in the program at all (Wang et al., 2016).

Thirdly, many types of ecosystem services are characterized by high levels of non-excludability (benefits cannot be fully captured by buyers). In these cases, individuals do not have direct incentives to pay for carbon sequestration, maintenance of water quality, or biodiversity conservation services generated by a remote forest because they can take a "free ride" as long as others pay for provision of the ecosystem service. Based on the same logic, individuals are reluctant to pay for the provision of ecosystem services knowing that some portion of the service flows will be captured by people who pay nothing. Thus the free-rider problem drives private WTP even lower (Champ et al., 2003; Freeman, 2003).

The fourth impediment to PES is the high transaction costs in ecosystem services trading (Stavins, 1995; Wunder et al., 2008). The so-called "Coase theorem"[1] showed that when there are clearly defined property rights and no transaction costs, valuating and trading externalities could result in socially optimal outcomes (Coase, 1960, 1988). But in reality, there are always transaction costs besides the production costs of ecosystem services provision, and in many cases high transaction costs become the largest barrier in the implementation of PES projects (Wunder et al., 2008). The major sources of transaction costs include: 1) measuring and validating ecosystem services; 2) costs in contract negotiations; 3) monitoring and enforcing ecosystem services provisions (Bromley, 1991; Wunder, 2005). High transaction costs make PES less attractive as a conservation approach, particularly when combined with other constraints of PES programs.

And lastly, friction derived from historical, organizational, and cultural factors in policy networks has been identified as an important impediment to implementation of PES (Wolf, 2013; Primmer et al., 2014). Creation and realization of incentive-based conservation schemes, as with any social intervention, is a process that occurs within an existing context and an existing set of social relation. PES may be perceived as threatening the knowledge, the justifications, and professional status of policy actors (Potter and Wolf, 2014). Therefore, incumbents occupying positions of authority in existing policy networks may constrain opportunities for institutional innovation.

To sum up, the fundamental reason for the underperformance of PES programs is the realization that the WTP of ecosystem services beneficiaries may not exceed the WTA of the providers plus transaction costs. In other words, investments from prospective buyers of ecosystem services are often insufficient to incentivize prospective sellers as well as cover substantial transaction costs (Wunder et al., 2008; Milder et al., 2010). While these constraints raise serious challenges, they also highlight potential new directions for PES research and application. We argue that the likelihood of realizing a functional PES scheme is expanded if practitioners can identify socioeconomic and ecological conditions that raise WTP of ecosystem services beneficiaries and lower WTA of the providers. In this article we use a simple economic model to analyze the demand and supply of ecosystem services in order to understand how wealth disparity between buyers and sellers shapes prospects for PES transactions. The economic model demonstrates that a certain level of wealth disparity between ecosystem services buyers and sellers can help elevate the WTP/WTA ratio and potentially overcome the barrier posed by transaction costs. Therefore, PES programs have a higher likelihood of success when established in contexts in which there is wealth disparity between buyers and sellers. Moreover, the economic model shows that when such wealth disparity exists, the high-income population is likely to contribute more in a PES program, while low-income population is likely to benefit more from the program. In such circumstances, PES can be an effective and socially progressive conservation strategy that advances both environmental and poverty alleviation objectives.

The rest of the article is organized as follows. Section 2 details the construction of the economic model, and Section 3 uses the model to analyze the relationship between wealth disparity and prospects for PES transactions. Section 4 addresses the research and policy implications of our findings and the associated limitations. Section 5 concludes with a discussion of future research directions.

2. Model Construction

In this model we assume that urban residents are the potential ecosystem services buyers, and private rural landowners are the potential ecosystem services providers. The utility function of the urban people is $u = u(x, q)$, with the budget restriction $I = p \cdot x + r \cdot q$, where x is the amount of market goods, p is price of market goods, q is the amount of ecosystem services which is generally fixed and non-rival in consumption, r is the rate charged for q, and I is income level. However, in most cases there is no direct charge for the public good q: for example, consumers do not typically pay for the level of ambient air quality, although they may incur additional expenses such as buying and operating air filters to ensure personal air quality levels. Hence, we will

[1] The "Coase Theorem" addressing contracting in a world of no transaction costs was not self-styled, but arose out of summaries of his work by other researchers, such as George Stigler. Coase himself explicitly disparaged the idea that transaction costs could be assumed to be negligible in a practical context (see Coase, 1988, p. 174–175).