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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).

assume that q is an unpriced public good, and that all income is spent on private goods.

A Cobb-Douglas utility function is used in this model, in the form of:

$$u = x^{\alpha} \cdot q^{1-\alpha} \tag{1}$$

in which $0<\alpha<1$, and α can be interpreted as the relative importance of ecosystem services in one's utility function. This utility function has the following property:

$$\frac{\partial u}{\partial x}>0,\ \frac{\partial u}{\partial q}>0,\ \frac{\partial^2 u}{\partial x \partial q}>0.$$

According to Hicks's welfare theory (Mas-Colell et al., 1995), WTP can be represented by compensating surplus (CS), which means the maximal value the buyer is willing to pay (WTP) to increase ecosystem services and maintain utility at the initial level. If the expenditure function for a consumer is $e = e(p, q, u)$, then CS can be written in the following form:

$$CS = e(p,\ q^0,\ u^0)\ -\ e(p,\ q^0+\Delta q,\ u^0) = I^o - e(p,\ q^0+\Delta q,\ u^0) \tag{2}$$

In which p is a vector of prices for private goods, q^0 is the original level of ecosystem services and Δq is the increase in ecosystem services after a PES program is implemented. Note that $e(p, q^0, u^0) = p \cdot x_0$ if x_0 is the solution for utility maximization problem. We assume that p remains constant, and is not affected by changes in q.

Let x_0 denote the original level of consumption of market goods, u^0 denotes the original utility level, and x_1 denote consumption of market goods after ecosystem services are increased but the utility level is maintained at u^0. Then we have:

$$u^0 = K \cdot {x_1}^{\alpha} \cdot (q^0+\Delta q)^{1-\alpha} = K \cdot {x_0}^{\alpha} \cdot (q^0)^{1-\alpha} \tag{3}$$

Solving this equation for x_1 we get:

$$x_1 = x_0 \cdot \left(\frac{q0}{q0+\Delta q}\right)^{\frac{1-\alpha}{\alpha}} \tag{4}$$

That is, given an increase in environmental services the amount of x needed to obtain the initial level of utility (u^0) will be lower, as expected. Inserting Eqs. (3) and (4) into Eq. (2) and rearranging, provides the following relationship:

$$WTP = \left(1-\left(\frac{q0}{q0+\Delta q}\right)^{\frac{1-\alpha}{\alpha}}\right) \cdot I^o \tag{5}$$

In this equation there are three parameters that determine WTP: initial income level I^o, the relative importance of ecosystem services in one's utility function α, and the ratio of $\Delta q/q^0$.

Rural people are assumed here to be small landowners, and their income from land is i. The landowners could quit agriculture or grazing activities and retire their land in order to provide ecosystem services, and they could find alternative jobs with alternative income $\beta \cdot i$, where β is the ratio of alternative income and original income.

The landowners choose to participate in PES projects if the payment P is greater than or equal to original income minus alternative income, so the condition for participation is:

$$P >= (1-\beta) \cdot i. \tag{6}$$

In other words, the minimum compensation required is:

$$WTA = (1-\beta) \cdot i \tag{7}$$

In equilibrium, P will equal marginal WTA and WTP. Define income density function for urban residents: $\varphi_1(I)$ and for rural residents: $\varphi_2(i)$. For simplicity of analysis, assume that each seller provides constant units of ecosystem services, E_1, and each beneficiary buys constant units of ecosystem services, E_2. Then, the aggregate demand function for ecosystem services is:

$$D = \int_{I(P)}^{\infty} E_1 \cdot \varphi 1(I) dI \tag{8}$$

The aggregate supply function is:

$$S = \int_{0}^{i(P)} E_2 \cdot \varphi 2(i) di \tag{9}$$

3. Model Analyses and Results

According to Eqs. (8) and (9), if we know the income density functions $\varphi_1(I)$ for urban residents and $\varphi_2(i)$ for rural residents, we can calculate the price per unit of ecosystem services and the quantity of ecosystem service units traded in equilibrium. In this section we assume specific distributions of wealth for each population, and run simulations to see how different values of the parameters could change WTP and WTA, thus changing the potential welfare gain from PES projects.

There are four factors that could change WTP and WTA: wealth distribution within population, which determines the forms of $\varphi_1(I)$ and $\varphi_2(i)$; the relative importance of ecosystem services in one's utility function, α; the ratio of $\Delta q/q^0$; and the ratio of alternative income and original income of the rural residents, β. To simplify the analysis, we assume that there is a uniform distribution of population density on a continuum of income levels. This means if the range of income in the whole population is [a, b], then for any point between a and b, the population density is the same. We first set α, β, and $\Delta q/q^0$ as constants to see the effects of wealth disparity on the welfare gain from PES projects.

Fig. 1 shows the welfare gain from PES projects in four different scenarios. Here we use the Gini coefficient as the indicator of the extent of wealth disparity. While the total income is the same in all four graphs, (a) and (c) have approximately the same Gini coefficient of 0.32, while (b) and (d) have the same Gini coefficient of 0.16, meaning that graphs (a) and (c) characterize settings with greater income inequality than graphs (b) and (d). Further in (a) and (b) it is assumed that there is regional wealth disparity, which means that the wealthier half of population live in urban areas and are ecosystem service buyers, while the poorer half live in rural areas and are sellers. In (c) and (d) it is assumed that there is no regional wealth disparity, which means there are rich and poor people in both urban and rural areas. The downward sloping curves represent demand of ecosystem services, while the upward sloping curves represent the supply. We use environmental surplus, a concept similar to net economic benefits, to measure the welfare gains from PES projects. We define environmental surplus (the green shaded areas) of a PES project as the triangle formed by the vertical axis and the supply and demand curves in each graph. From Fig. 1 we can see that the Gini coefficient, the indicator of wealth disparity, has a significant influence on the amount of environmental surplus: the larger the Gini coefficient is (in (a) and (c)), the more the environmental surplus could be achieved from a PES project. Besides the influence of Gini coefficient in the whole population, regional wealth distribution is also an important factor: the scenarios with regional wealth disparity (in (a) and (b)) have larger environmental surplus from PES than those without regional wealth disparity (in (c) and (d)).

This model could also be used to simulate scenarios with different α, $\Delta q/q0$, β, and transaction costs. According to Eqs. (5) and (7):

$$\frac{\partial WTP}{\partial \alpha}<0, \quad \frac{\partial WTP}{\partial\left(\frac{\Delta q}{q0}\right)}<0, \quad and \frac{\partial WTA}{\partial \beta}<0.$$

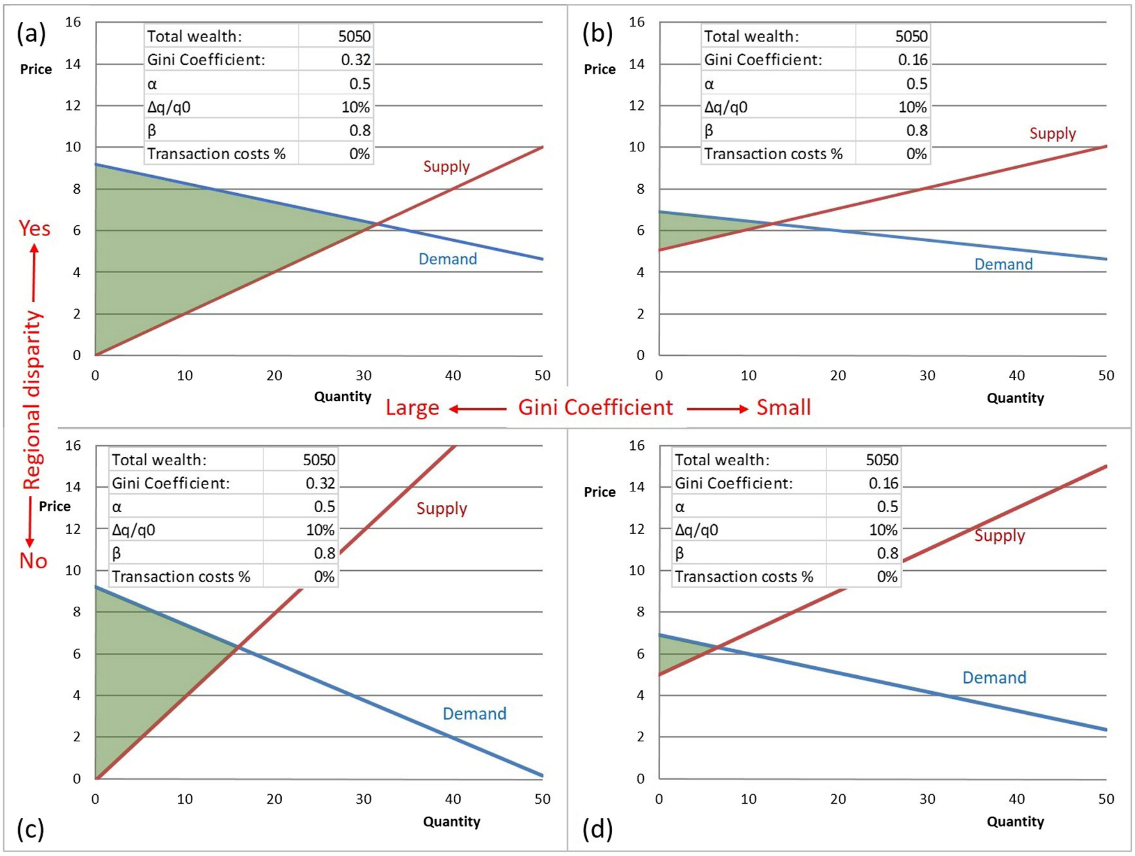

Fig. 1. Environmental surplus of PES projects in different scenarios.

Therefore, letting the original equilibrium be denoted by E_0, larger $\Delta q/q0$ will lead to higher WTP and move E_0 to E_1, and larger β will lead to lower WTA and move E_0 to E_2, both of which will increase the environmental surplus of PES projects. In contrast, larger α will lead to lower WTP and move E_0 to E_3, reducing the potential environmental services. Transaction costs can also be introduced into this framework, in effect leading to higher WTA and moving from E_0 to E_4.[2] Such costs are expected to decrease the realized environmental surplus of PES projects. Fig. 2 shows the shift of demand and supply curves due to changes in α, $\Delta q/q^0$, β, and transaction costs.

Results of the model simulations above identify the favorable conditions for PES programs to be effective. First, higher wealth disparity, particularly regional wealth disparity between ecosystem service buyers and sellers, could increase the chances of transactions. This is because when all other conditions are equal, on the one hand, high income populations are likely to have higher willingness to pay, because their spending on ecosystem services is a relatively small fraction of their income; on the other hand, low income population are likely to be willing to accept lower payments, because their opportunity costs for ecosystem service provision are relatively low. Secondly, environmental attitudes and the knowledge and awareness of the significance of ecosystem services (α in the model) play an important role in PES. Societies with high awareness of the values of ecosystem services are more likely to be successful in PES programs. Thirdly, WTP has positive correlation with expectation of the PES programs ($\Delta q/q^0$ in the model): the buyers are willing to pay more if they expect that the PES programs could significantly improve provision. Fourthly, suppliers are more willing to participate in PES programs and accept lower payments if they have alternative income sources (β in the model) that could largely offset losses associated with ecosystem service provision. Last but not least, reducing transaction cost to a reasonable level is key to the success of a PES program.

4. Discussion

PES is a market-based policy instrument that has potential to address complex environmental issues, particularly when private landowners are involved. But the overwhelmingly low ratio of WTP to WTA and transaction costs undermine the effectiveness of PES schemes in many real world situations. The frustrations that have grown out of

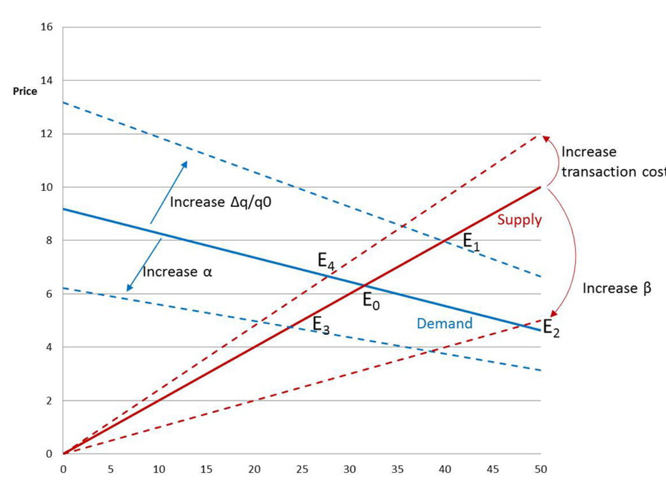

Fig. 2. Shifting demand and supply curves due to changes in α, $\Delta q/q^0$, β, and transaction cost.

2 Here we assume, without loss of generality, that the transaction cost is represented on the supply side and the marginal transaction cost is constant.

the numerous pilot PES programs indicate that PES is not a universally applicable conservation tool as many people have promoted (see Silvertown, 2015). There is a need for a more targeted approach to the design and application of PES. The economic model proposed in this article helps identify the favorable conditions for PES schemes to overcome the obstacles and to focus attention on the regions and populations with high potential to realize PES programs. The model shows that, under the general assumption that relatively higher income populations tend to express higher willingness to pay for ecosystem services while lower income populations tend to express lower willingness to accept payments for ecosystem service provision, PES programs have a higher likelihood of success when established in or across regions where there is wealth disparity between buyers and sellers. *Ceteris paribus*, the potential for welfare gains form a PES program are enhanced by wealth disparities between buyers and sellers.

The economic model also indicates that a PES program could not only be ecologically effective and economically viable, but also socially progressive. This is because when wealth disparity exists between the buyers and sellers of PES, the wealthier population would contribute more into the program, while poorer population would benefit more from it. Thus the model also provides important justifications for integration of PES and poverty alleviation programs. Some researchers argue that PES programs should not include poverty alleviation objectives, since they would distract the focus of the programs from environmental improvement (Engel et al., 2008; Gauvin et al., 2010; Milder et al., 2010). But we argue that because of the high WTA of ecosystem service sellers, the buyers need to pay the sellers compensation in excess of the market value of the sellers' actual loss in order to incentivize ecosystem service provision. This 'over-payment' could be justified if the sellers are economically disadvantaged and need external help to overcome the economic hardship. In other words, poverty alleviation objectives could help PES programs gather more political and financial support. This explains why most real-world PES schemes, such as the Slope Land Conversion program in China (Bennett, 2008) and the REDD+ program between developed and developing countries (Corbera and Schroeder, 2011), have explicit poverty alleviation objectives.

Our model is applicable in situations in which the poor have secure land tenure, but the productivity of their land and enterprises is relatively low. It is important to note that if the poor are landless, or their tenure is insecure, PES may aggravate income disparities. The poor could be ineligible to participate in PES, or could be displaced from the land when more powerful actors take control of the land to capture increased value (Fairhead et al., 2012; Osborne, 2013). In such contexts, PES is not a suitable conservation tool from a social equity perspective, and other types of policy instruments should be considered.

Also, our model assumes voluntary transaction on the sides of both PES buyers and sellers. In practice, a free market for ecosystem services is not likely to emerge because of the varied constraints we have specified. Therefore, PES programs often arise from regulation, such as a cap-and-trade system for CO_2 emissions, or the government serves as the sole buyer as is the case in China's SLCP. In these cases, the government could impose taxes on certain groups and use the money to elicit provision of ecosystem services. While the demand side of PES is not strictly voluntary in such circumstances, our model is still valuable for highlighting how relative wealth of beneficiaries and sellers shapes the economics and politics of PES schemes. Before establishing a PES project, it is important to examine if there is a relatively high income region/population with high willingness to pay, and a relatively low income region/population with potential to provide ecosystem services and high willingness to participate, even if payments are made by the government on behalf of the beneficiaries. The government's involvement in PES schemes can also facilitate research and practice that advance specification and accounting standards for ecosystem services. It is possible to imagine that government acting in the role of buyer in the initial period can incubate standards and learning that later on support expanded private exchange.

Results of our model simulations raise a series of empirical questions that invite future research. First, is there a threshold (or minimum) wealth disparity that must be crossed in order to enable meaningful PES transactions and to realize poverty alleviation, and what contextual variables shape this threshold? Second, what groups of citizens are the actual beneficiaries of governmental purchases of ecosystem services, and to what extent do they overlap with the groups on whom taxes and fees are levied? Third, in what contexts can PES function as a primary poverty alleviation strategy, and how can it complement other development policies? Responses to these questions will help us better understand the equity implications of PES as a tool for environmental conservation and economic development.

5. Conclusion

Despite the economic rationale and some enthusiasm for integrating ecosystem services into environmental management, efforts to roll out PES have largely met with frustration. Researchers and practitioners have found it difficult to demonstrate the validity of the original conception of PES that focused on free-market exchanges. PES is now increasingly understood as an open and flexible concept that involves different types of incentive-based policy instruments. The success of PES, therefore, relies not only on market mechanisms, but also on socioeconomic and institutional conditions that make it possible to overcome these constraints. Under the newly relaxed specification of PES, the identification of such conditions becomes one of the key tasks for PES researchers and practitioners.

Our study makes a contribution by highlighting one important favorable condition for PES: wealth disparity between ecosystem services buyer and sellers. We do not regard this structural factor as necessary or sufficient, but we believe it is important for understanding performance of PES and for assessing the suitability and design of PES. The results of our analysis speak to a pragmatic approach to advancing environmental conservation and economic welfare. These insights can help policy makers and environmental management professionals to target PES programs toward suitable regions and populations, thereby increasing chances for success. After all, ecosystem services are not ordinary market goods, so their exchange demands unconventional market mechanisms.

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