EXPLORING THE EFFICIENCY AND EFFECTIVENESS OF PAYMENTS FOR ECOSYSTEM SERVICES IN CHINA'S WOLONG NATURE RESERVE

By

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ABSTRACT

EXPLORING THE EFFICIENCY AND EFFECTIVENESS OF PAYMENTS FOR ECOSYSTEM SERVICES IN CHINA'S WOLONG NATURE RESERVE

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Conservation investments, including payments for ecosystem services (PES), have been increasingly devoted to protecting and restoring ecosystems. However, the efficiency and effectiveness of PES programs depend on the program design, biological, economic and social conditions, and dynamic trends in population and households. This dissertation focused on the efficiency and effectiveness of payments for ecosystem services. My study area is China's Wolong Nature Reserve for giant pandas where human interact with the environment under the Grain-to-Green program (GTGP) and the Natural Forest Conservation program (NFCP). Specific objectives for this dissertation are to: (1) evaluate the effects of social and economic factors on the reenrollment in the GTGP after the current contracts expired, (2) target land for enrollment in the GTGP, (3) assess the impacts of social capital and labor migration on the fuelwood consumption of indigenous people, and (4) model the impacts of dynamics in population and households on the effects of the NFCP. This research use interdisciplinary methods and tools for human-environment interactions in a coupled human and natural system (CHANS).

I used stated choice method with main effects design to measure the effects of conservation payment, social norms and program duration on the reenrollment in the GTGP, controlling for household characteristics and land features. In addition to conservation payment amounts and program duration, social norms had significant impacts on program reenrollment. Farming income had a negative effect on program

reenrollment, while income from rural-urban labor migrants had a positive effect on program reenrollment. I then explored cost-effective targeting of land for enrollment in the GTGP. Environmental benefits of lands and opportunity costs for land enrollment were estimated using land features and household characteristics. The efficiency of investments in a discriminative payment scheme (payments differ according to opportunity costs) was substantially higher than in a flat payment scheme (same price paid to all participants). In addition, both optimal targeting and suboptimal targeting achieved substantially more environmental benefits than random selection of land.

To assess the impacts of social capital and labor migration on the fuelwood consumption, I used propensity score techniques. Results suggested that social capital in the form of weak social ties to people in urban settings had significant impacts on rural-urban labor migration. Following the chain of capital substitutions, labor migration increased household income, which in turn reduced fuelwood consumption. Simulation results from the systems model suggested substantial panda habitat can be obtained from both cash payment and electricity payment scenarios. However, electricity payment, as a more direct payment approach for reducing human impacts, can improve the efficiency of conservation investments. The effects of conservation investments are non-linear due to increases in human population and number of households. In addition, policy effects can be uncertain due to uncertainties in the behavior of new households that have not been included in the NFCP. The approach and methods in this research may also be applied in many other CHANS.

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CHAPTER 1

INTRODUCTION

1.1 BACKGROUND

Much of the world's natural land cover has been transformed by human activities (Foley et al. 2005, Morton et al. 2006), resulting in ecosystem degradation and biodiversity loss worldwide (Vitousek et al. 1997, Green et al. 2005). Human alteration of land cover is not limited to human-dominated areas, as it is also very common in many of the world's protected areas, such as nature reserves (Liu et al. 2001, Curran et al. 2004). Interventions have been used to counter this trend through development activities such as stimulating community economies (e.g., ecotourism), encouraging community-based natural resources management, providing social benefits (e.g., education), and redirecting labor and capital from activities that harm ecosystems. These indirect approaches have been referred to as "conservation by distraction" (Ferraro 2001, Ferraro and Simpson 2002). Although billions of dollars have been invested by governments, private sectors, and conservation non-government organizations (NGOs) through these approaches, the deterioration of ecosystems continues (James et al. 2001, Ferraro and Kiss 2002, Fearnside 2005).

One approach to improve the efficiency of conservation investments is through Payments for Ecosystem Services (PES) that provide incentives directly to ecosystem services providers to undertake actions for desired environmental benefits (Ferraro and Kiss 2002, Wunder 2007, Jack et al. 2008). In recent years, PES programs, including land set aside and forestry contracting, have been implemented in many countries (Bennett 2008, Claassen et al. 2008, Wunder 2008). Since current conservation investments are far below the requirements for conserving ecosystems globally (James et al. 1999, James et al. 2001), the efficiency and effectiveness of conservation investments in these PES programs have been a great concern to conservation practitioners.

From the perspective of policy design, many PES programs are short-term, compared to outright purchases, with uncertainties in land use after the programs end (Claassen et al. 2008). One approach to sustain the conservation gains from PES programs is through continued conservation investments. In addition, allocation of scarce conservation funds is critically important. Previous studies suggested that the efficiency of conservation investments is affected by heterogeneous biological values, demographic, political and socioeconomic conditions (O'Connor et al. 2003, Polasky et al. 2005, Wilson et al. 2006). Based on biological values and human impacts, 25 regions have been identified as global conservation priority (Myers et al. 2000). However, heterogeneous social contexts (e.g., social norms) have often been neglected when allocating conservation resources. Efficient conservation investments, therefore, need take heterogeneous human and environmental factors into consideration. Sustainable conservation gains can be achieved only if participants would maintain these conservation gains even after PES programs end (Uchida et al. 2005). For example, rural-urban labor migration may not only lower farmers' dependence on the ecosystems that are targeted for conservation, but also reduce human impacts on the places of origin. Moreover, both human and natural systems are not isolated. The effectiveness of conservation investments may largely depend on the dynamic interactions among human and natural components.

Studies on the efficiency and effectiveness of PES programs need to integrate human society with natural environment and use methods from both social and natural sciences due to complex interactions among components in coupled human and natural systems (Liu et al. 2007a, Liu et al. 2007b). This dissertation attempts to address some of these critical issues using China's two PES programs in Wolong Nature Reserve. Understanding of these issues will not only help improve the efficiency and effectiveness of existing PES programs, but also will have profound implications for the allocation, design, and implementation of future conservation investments such as the United Nation's collaborative initiative on Reducing Emissions from Deforestation and forest Degradation (REDD) in developing countries.

1.1.1 PES programs around the world

The most famous PES program is probably the Conservation Reserve Program (CRP) in the United States. The CRP is a subtitle of the Conservation Title of the Food Security Act of 1985. The main objective of the CRP was to reduce soil erosion caused by agricultural production, with secondary objectives of creating wildlife habitat, improving water quality, controlling crop supply, and transferring income to farmers (Smith 1995, Johnson et al. 1997). Enrolled farmers receive conservation payments for converting highly erodible or environmentally sensitive cropland to grass, trees, or other conservation uses through a contract that typically lasts 10 years. From 1986 to 1992, about 36.4 million acres of cropland were enrolled in the CRP at an average annual payment of $50 per acre (Skaggs et al. 1994, Cooper and Osborn 1998), which resulted in a cost of over $1 billion per year (Parks and Schorr 1997). By the end of 2005, 35.9 million acres of land was enrolled in the CRP with an annual cost of approximately $1.8 billion (Claassen et al. 2008).

Some other developed countries have also implemented ambitious PES programs (OECD 1997). The Permanent Cover Program (PCP) in Canada began in 1989. The PCP was intended to conserve and improve soil productivity by retiring cropland where annual cultivation was causing long-term soil damage and to generate benefits for water, wildlife habitat, and landscapes. About 1.3 million acres of marginal and erodible cropland was converted from grain production to pasture or forests. In return, enrolled farmers received one-time conservation payments of $15 and $22 per acre for 10-year contracts or $36 and $47 per acre for 21-year contracts for pasture and forests, respectively. The total cost of the PCP was around $51 million. The European Union (EU) also has implemented conservation payment programs for agricultural land conversion. As part of the reforms of the Common Agricultural Policy, two land-conversion programs were introduced in 1992. The first one is part of the agri-environmental regulation, a set of policies aimed to promote agricultural production compatible with protection of the environment. Enrolled farmers receive an annual conservation payment of up to $784 per ha for setting aside agricultural land for at least 20 years to prevent soil erosion and improve water quality. Specific implementations in member countries are different due to the diversity of environmental conditions and agricultural structures. Another program of the EU is an afforestation scheme, which pays for afforestation of agricultural land to reduce the wood shortage in the EU. Enrolled farmers receive a payment covering the cost of afforestation and new woodland maintenance along with an annual conservation payment of up to $947 per ha for up to 20 years. By 1997, the afforestation scheme had converted around 930 000 ha of land at a cost of about $2.6 billion.

In the developing world, Costa Rica's Pagos de Servicios Ambientales (PSA) program provides a well-known example of a payment for ecosystem service program. Since 1997, the PSA has been implemented with three subprograms: reforestation, sustainable forest management, and forest conservation (Zbinden and Lee 2005, Sierra and Russman 2006, Pagiola 2008). The reforestation subprogram subsidizes conversion of cropland to forest for 15 years. Enrolled farmers have to maintain a tree survival rate of at least 85% to receive a total payment of approximately $550 per ha. In the sustainable forest management subprogram, only valuable trees beyond a threshold diameter are allowed to be cut during a contract of 10 years. As compensation, enrolled forest owners receive a payment of approximately $327 per ha. Moreover, access roads to forest plots are limited to reduce the disturbance due to timber harvesting. The forest conservation subprogram rents forest land from owners for 5 years with a payment of approximately $210 per ha, and enrolled owners were not allowed to harvest timber or develop the land for other uses (e.g., livestock breeding) during the contract. By 2001, the PSA had provided conservation payments to more than 4400 farmers and forest owners, and the total area of land enrolled in the PSA was more than 284 000 ha, which is about 5.5% of Costa Rica's national territory (Zbinden and Lee 2005). Many other ecosystem protection programs have also recently been launched at regional and national levels, such as the payment for forest protection programs in Los Negros of Bolivia and Pimampiro of Ecuador (Asquith et al. 2008, Wunder 2008, Wunder and Alban 2008) and Payment for Hydrological Environmental Services program (Pago de Servicios Ambientales Hidrológicos, PSAH) in Mexico (Munoz-Pina et al. 2008).

In addition to their main objectives of converting land cover and protecting forests (Smith 1995, Zbinden and Lee 2005), many of these programs have also achieved objectives in conserving/creating wildlife habitat and in restoring ecosystems (Dunn et al. 1993, Johnson and Schwartz 1993, McMaster and Davis 2001, Sierra and Russman 2006, Asquith et al. 2008). However, there may be uncertainties in land use after these programs end. For example, studies of the CRP in the United States have shown that most of the enrolled land (60% or higher) is likely to be reconverted to crop production when contracts end (Johnson et al. 1997, Cooper and Osborn 1998). Since 1996, expired CRP contract holders could apply for re-enrollment. Even with continued payments with similar prices, only about 55% of previous CRP land was re-enrolled in new contracts by the end of 2001 (Claassen et al. 2008).

1.1.2 PES programs in China

Over the past three decades, China's economy has grown faster than any major nations, fueling unprecedented ecosystem degradation that has caused devastating socioeconomic impacts (Liu and Diamond 2005). For instance, the severe droughts in 1997 and the major floods in 1998 has been recognized at least partially as a result of farming on steep slopes and excessive deforestation (World Bank 2001). To mitigate the impacts of the degraded ecosystems, China has been implementing two nation-wide PES programs, the Grain-to-Green Program (GTGP, also referred to as the Sloping Land Conversion Program) and the Natural Forest Conservation Program (NFCP, also referred to as the Natural Forest Protection Program) (Xu et al. 2006, Liu et al. 2008). The GTGP converts sloping cropland to forest or grassland by providing participating farmers with conservation payments, whereas the NFCP protects natural forests through logging bans and afforestation by providing incentives to forest enterprises and rural communities.

The GTGP has been implemented since 1999. Due to its main objective of reducing soil erosion by increasing vegetative cover, the criterion for land conversion in the GTGP is for the slope of cropland in southwestern China to be >25° and cropland in northwestern China to be >15°. Although cropland with slopes above the threshold receives priority in enrollment, many cropland plots with slopes below the threshold can also be enrolled (Uchida et al. 2005). Participating farmers receive conservation payment for a maximum of 8 years. The government offers farmers an annual payment of 2250 kg and 1500 kg of grain or cash payments of 3150 and 2100 yuan per ha (as of June, 2010, 1 USD = 6.8 yuan) of enrolled cropland in the upper reaches of the Yangtze river basin and in the middle-upper reaches of the Yellow river basin, respectively. In addition, annual miscellaneous expenses of 300 yuan per ha and a one-time subsidy of 750 yuan per ha for seeds or seedlings were provided. By the end of 2006, the GTGP had converted about 9 million ha of cropland (Liu et al. 2008). Studies have shown that the GTGP has substantially improved ecosystem services such as increased forest cover, reduced water surface runoff and soil erosion, reduced river sediments and nutrient loss for maintaining soil fertility, and reduced desertification (Liu et al. 2002, Ma and Fan 2005, Li et al. 2006, Liang et al. 2006, Long et al. 2006, Xu et al. 2006, Wang et al. 2007). While these conservation gains are encouraging, the cost of the GTGP is also tremendous. By the end of 2005, over

90 billion yuan had been invested in the GTGP, and it is expected that the total investment in the GTGP will reach 220 billion yuan by 2010 (Liu et al. 2008).

The NFCP has been implemented as a pilot program in 1998 with full implementation since 2000. The aims of the NFCP are to (1) protect and restore natural forests through logging bans in the upper reaches of the Yangtze river basin and the middle-upper reaches of the Yellow river basin by 2000 and reduction in harvesting elsewhere; (2) construct plantation forests through aerial seeding and artificial planting to increase the capacity for timber harvesting from plantation forests; (3) create alternative employment for traditional forest enterprises (Zuo 2002a, Liu et al. 2008). The NFCP planned to reduce timber harvests in natural forests from 32 million m^3 in 1997 to 12 million m^3 in 2003, and afforest 31 million ha by 2010. Conservation payments are provided for 10 years. Specific payments for different conservation actions are 750 yuan/ha for aerial seeding; 1,050 yuan/ha for forest regeneration through mountain closure; 3,000 and 4,500 yuan/ha for artificial planting in the Yangtze and Yellow river basins, respectively; and 10,000 yuan per worker for protecting 340 ha of forests (Xu et al. 2006). By the end of 2005, about 61 billion yuan has been invested through the NFCP (Liu et al. 2008). Although the NFCP has been shown to have produced substantial ecosystem services such as increased carbon sequestration (Hu and Liu 2006) and reduced soil erosion (Zhang 2006), China's timber imports from other countries has increased at least in part due to the NFCP (State Forestry Administration of China 2005-2007).

Most of the GTGP contracts matured in 2008. To sustain the conservation gains from the GTGP, the program was extended for another cycle of up to 8 years. It is expected that the conservation payments under the NFCP will also be extended when the initial contracts end in 2010. However, another cycle of these programs may not guarantee the sustainability of their conservation achievements into the future. For instance, some land enrolled in the GTGP may be reconverted to agriculture when the payments cease (Uchida et al. 2005). These two programs may provide especially important opportunities to avoid deforestation and restore degraded ecosystems in many biologically significant regions such as nature reserves (Loucks et al. 2001). Given the tremendous investments in these two programs, it is important to evaluate the efficiency of conservation investments in these programs and the effects of these programs on ecosystem recovery.

1.2 RESEARCH OBJECTIVES

The overall goal of this dissertation is to explore the efficiency and effectiveness of PES programs in China's Wolong Nature Reserve. In Chapter 2, I evaluated the effects of social norms, together with other household level socioeconomic factors and land features, on the reenrollment of lands that have been enrolled in the GTGP after the current contracts end. In Chapter 3, I demonstrated the environmental benefits that can be obtained through cost-effective targeting of lands that have been enrolled in the GTGP using household characteristics and land features under different payment schemes. In addition to the direct effects from the GTGP, such as increased forest cover, the GTGP may also produce indirect effects on the environment. Studies on the GTGP found that part of the labor force has been released from agriculture and has boosted the trend of rural-urban labor migration (Bao et al. 2005, Liu 2005, Ge et al. 2006, Hu et al. 2006, Uchida et al. 2009); hence human population pressure on the ecosystem has been reduced (Liu et al. 2007a). In Chapter 4, I assessed the impacts of social capital and rural-urban labor migration on fuelwood consumption. By linking social capital to labor migration and then to the use of fuelwood (one form of natural capital), I examined the substitution among different forms of capitals for sustainability. For Chapter 5, I developed a spatially explicit model to study the effects of the NFCP and alternative policy scenarios and dynamics in population and households on panda habitat. In Chapter 6, I summarized the results of previous chapters and discuss their implications for policies that aim to achieving sustainability. Specific objectives include:

1. Linking social norms to efficient conservation investment in payments for ecosystem services (Chapter 2)
2. Using cost-effective targeting to enhance the efficiency of conservation investment in payments for ecosystem services (Chapter 3)
3. Understanding the impacts of weak ties and labor migration on the environment (Chapter 4)
4. Modeling the effects of payments for ecosystem services in a coupled human-nature system (Chapter 5)

1.3 STUDY AREA

Wolong Nature Reserve was established in 1963 with an area of 200 km^2 and was expanded to 2000 km^2 in 1975 (Figure 1.1). Located in southwest China, within one of the 25 global biodiversity hotspots (Myers et al. 2000), Wolong Nature Reserve is one of the largest reserves for the protection of endangered giant pandas (*Ailuropoda melanoleuca*). The wild panda population in the reserve represents about

10% of wild pandas in China, the only country with wild pandas in the world. In addition to bamboo, which is the pandas' main diet, conifer and broadleaf forests are important components of panda habitat by providing shelter and cover (Schaller et al. 1985). Wolong Nature Reserve also provides habitat to more than 6,000 plant and animal species. As a coupled human and natural system (Liu et al. 2007b), the reserve is also home to about 4,500 human residents in about 1200 households distributed between two townships (Wolong and Gengda). People in the reserve engage in diverse economic activities such as fuelwood collection, deforestation for agricultural land, road construction, and supporting tourism. Commercial timber harvesting has been prohibited in the reserve. Previous studies in this reserve have demonstrated rapid degradation in panda habitat due to these human impacts, especially fuelwood collection and deforestation for agricultural land (An et al. 2005, Viña et al. 2007). Almost all of these impacts were within 6 km (Figure 1.1) of households (Linderman et al. 2005). Moreover, the rapidly increasing human population and even more rapid increase in the number of households have produced increasing human impacts on the ecosystem in the reserve (Liu et al. 2003a).

Although enormous amounts of time and energy were needed for fuelwood collection due to extremely rugged terrain and the difficulty of fuelwood collection was increasing due to shrinks in forested area, indigenous people in the reserve still rely much of their energy requirements on fuelwood for cooking and heating. The reserve administration had limited the amount of fuelwood collection, however, it was difficult for the administration to monitor and enforce due to the complex terrain and broad scale of the reserve (Figure 1.1). Electricity, as an alternative to fuelwood, is available in Wolong, but it had been used mainly for lighting and some electronic appliances (e.g., TV) because electricity was expensive for them to afford, and the voltage and stability of electricity were not reliable (An et al. 2002). To encourage the use of electricity as a replacement of fuelwood, the electricity networks in Wolong were reconstructed in 2001 leading to greatly improved voltage and stability of electricity. However, these conservation policies and efforts were not effective without providing conservation payments to indigenous people or substantial investments in monitoring (Liu et al. 2007a).

The GTGP enrollment took place in Wolong Nature Reserve in 2000, 2001, and 2003. Farmers were encouraged to enroll their cropland plots with slopes over 25 degrees, but many cropland plots with slopes below 25 degrees were also allowed to enroll. The GTGP may generate a number of positive impacts for the protection of panda habitat in Wolong Nature Reserve. The most immediate observable impact, for instance, is that part of the labor force has been released from agriculture and has boosted the trend of rural-urban labor migration. In the long run, degraded panda habitat may recover because the GTGP increased forest cover, which is an important component of panda habitat (Liu et al. 1999, Liu et al. 2001). In addition, GTGP land may generate substantial fuelwood, therefore alleviating further degradation of panda habitat due to fuelwood collection in natural forests. However, removal of trees for fuelwood from GTGP land may compromise the GTGP's potential for restoring panda habitat.

The NFCP enrollment took place in Wolong Nature Reserve in 2000. All households that existed in 2000 were enrolled in the NFCP for 10 years. No addi

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additional enrollment of NFCP has been offered after 2000, and new households (households formed after 2000) were not included in the program. Under the NFCP contract, every 5~10 households have been allocated a natural forest parcel to monitor to prevent illegal harvesting. Illegal harvesting refers to cutting trees in natural forests. However, people are allowed collect branches of trees near their households. Each participating household is provided an annual payment of about 850 yuan, which accounts for 14% of average annual income of households in Wolong in 2001 (He 2008). If illegal timber harvesting is found in a natural forest parcel, the monitoring households will lose part or all of their NFCP payment of the year depending on how heavy the parcel gets harvested. Illegal harvesters, if get caught, will lose their NFCP contracts. Participating households are encouraged to use the payment to purchase electricity to replace fuelwood for conservation.

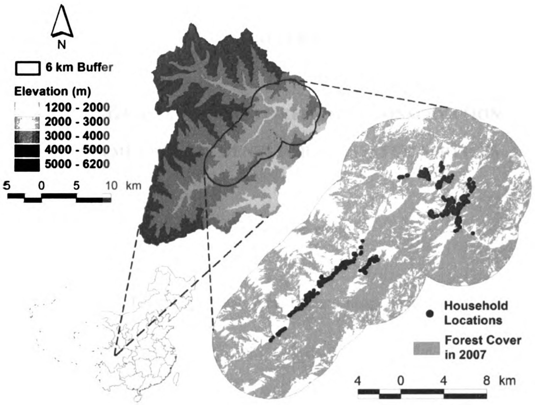

Figure 1.1. Locations and elevations of Wolong Nature Reserve, China, and households in the reserve.

CHAPTER 2

LINKING SOCIAL NORMS TO EFFICIENT CONSERVATION INVESTMENT IN PAYMENTS FOR ECOSYSTEM SERVICES

In collaboration with

Frank Lupi, Guangming He, and Jianguo Liu

ABSTRACT

An increasing amount of investment has been devoted to protecting and restoring ecosystem services worldwide. The efficiency of conservation investments, including payments for ecosystem services (PES), has been found to be affected by biological, political, economic, demographic, and social factors, but little is known about the effects of social norms at the neighborhood level. As a first attempt to quantify the effects of social norms, we studied the effects of a series of possible factors on people's intentions of maintaining forest on their Grain-to-Green Program (GTGP) land plots if the program ends. GTGP is one of the world's largest PES programs and plays an important role in global conservation efforts. Our study was conducted in China's Wolong Nature Reserve, home to the world-famous endangered giant pandas and more than 4,500 farmers. We found that in addition to conservation payment amounts and program duration, social norms at the neighborhood level had significant impacts on program re-enrollment, suggesting that social norms can be used to leverage participation to enhance the sustainability of conservation benefits from PES programs. Moreover, our results demonstrate that economic and demographic trends also have profound implications for sustainable conservation. Thus, social norms should be incorporated with economic and demographic trends for efficient conservation investments.

2.1 INTRODUCTION

Current investments are far below the requirements for conserving ecosystems globally (James et al. 1999, James et al. 2001). Moreover, most of these investments are spent within wealthy countries, whereas places with rich biodiversity under threat are often poor (James et al. 1999, Brooks et al. 2006). To minimize biodiversity loss with limited conservation resources, priorities for conservation investments have been placed on areas where biodiversity and human impacts are highest, e.g., global biodiversity hotspots (Mittermeier et al. 1998, Myers et al. 2000, Brooks et al. 2006). However, priority settings based on biological values and threats to these values alone may not guarantee the efficiency of conservation investments.

Efficient conservation investments need to incorporate biological values with heterogeneous demographic, political, and socioeconomic conditions (O'Connor et al. 2003, Polasky et al. 2005, Wilson et al. 2006). The high human population and household density and growth rates in the biodiversity hotspots indicate that human population is and will remain an important factor in global biodiversity conservation (Cincotta et al. 2000, Liu et al. 2003a), and the uneven distribution of human population and households should be considered in conservation investments (Luck et al. 2004). Political conditions (e.g., political corruption and government stability) in targeted regions also have a pronounced effect on the efficiency of conservation investments (Smith et al. 2003). Like human population, per unit area costs of effective conservation also vary enormously across different places (Balmford et al. 2003). The efficiency of conservation investments can be improved by considering economic conditions, such as land prices, at global, regional, and local scales (Ando et al. 1998, Balmford et al. 2000, Odling-Smee 2005, Armsworth et al. 2006). Although much has been learned about the effects of these socioeconomic factors on the efficiency of conservation investments (Ando et al. 1998, Balmford et al. 2000, Cincotta et al. 2000, Balmford et al. 2003, Liu et al. 2003a, O'Connor et al. 2003, Smith et al. 2003, Odling-Smee 2005, Wilson et al. 2006), little is known about the effects of social norms at the neighborhood level (Ehrlich and Levin 2005).

Social norms are shared understandings of how individual members should behave in a community under a given circumstance, and members within the community reward or punish people for their behaviors in following or breaking the norms (Coleman 1990, Bendor and Swistak 2001). More generally, social norms may also be sustained by the feelings of reputation and self-esteem through conforming to social norms, or shame and guilt through detachment from the norms even in the absence of third-party punishment (Elster 1989, Coleman 1990, Cialdini and Goldstein 2004). In this chapter, we study social norms in the more general context, which may be sustained by both self-enforced psychological feelings and/or third-party enforced punishment. Specifically, we examine when an individual's behavior is directly influenced by the behavior of other members in the community, and substantial change in aggregate behavior of the community can change an individual's behavior (Manski 2000, Dietz 2002). Social norms also have been important in the collective actions of natural resources management (Sethi and Somanathan 1996, Ostrom 2000, Dietz et al. 2003, Pretty 2003) but have received little attention in studies of conservation investments.

One approach to conservation investments is through Payments for Ecosystem Services (PES) (Smith 1995, Daily 1997, Ferraro and Kiss 2002, Zbinden and Lee 2005), such as land set aside and forestry contracting in the United States and European Union (OECD 1997). In contrast to outright purchase of land or permanent easements, short-term PES programs may result in only temporary conservation benefits, with uncertainty about land use after the programs end. Past studies have focused on the program participation of landowners (Langpap 2004, Zbinden and Lee 2005), but much less is known about the impacts of subsequent policies on land use when a PES program ends. Subsequent PES programs are very important for the sustainability of conservation benefits from initial PES programs.

Past studies have suggested that post-program land use of farmers who participated in conservation payment programs can be determined by their sociodemographic conditions (Johnson et al. 1997, Cooper and Osborn 1998). Agricultural income has been shown to have positive impacts on the reconversion of enrolled land (Cooper and Osborn 1998). Furthermore, farmers tend to enroll marginal land into conservation programs (Zbinden and Lee 2005). In regard to the characteristics of respondents, older contract holders tend not to reconvert their enrolled land (Cooper and Osborn 1998).

People's decisions to participate in a PES program are made in a social context. Studies indicate that both economic incentives and social norms are important in an individual's behavior (Lindbeck 1997) in terms of common resources management (Levin 2006, Vincent 2007). Individuals whose land-use decisions differ from the majority in the community may be exposed to social pressures from the community. Studies of individuals' participation in PES programs have focused on the incentives provided by conservation payments (Smith 1995, Cooper and Osborn 1998); little is known about the impacts of social norms at the neighborhood level on the sustainability of conservation, although substantial conservation benefits (e.g., through land enrolled in conservation contracting programs) may be produced with a relatively small change in policy or other exogenous factors due to social norms (Lindbeck et al. 1999, Nyborg and Rege 2003). To illustrate the impacts of social norms on the sustainability of conservation, we studied the impacts of subsequent policies on the land plots that have been enrolled in the Grain-to-Green Program (GTGP) in China's Wolong Nature Reserve.

In this chapter, we focus on the re-enrollment intentions of local inhabitants regarding their GTGP land plots that are likely to be reconverted to agriculture when the program ends, given different PES policy scenarios following the GTGP. Our policy scenarios were combinations of three attributes: conservation payment, program duration, and neighbors' behavior (the percentage of neighbors reconverting their enrolled land plots to agriculture). We used stated-choice methods (Louviere et al. 2000, Naidoo and Adamowicz 2005) to relate these attributes to the re-enrollment of those GTGP land plots that are likely to be reconverted when the GTGP ends. In addition, controls were set for household economic and demographic conditions, features of the GTGP land plots, as well as characteristics of respondents.

2.2 METHODS

2.2.1 Household surveys

We conducted household surveys in Wolong from May to August of 2006. We chose household heads or their spouses as our interviewees because they are usually the decision makers of household affairs. Our questionnaire was iteratively pre-tested and revised using qualitative interviews with 54 randomly chosen local households (Presser et al. 2004). The finalized survey was implemented on a sample of 321 households, which represent ≈26.8% of households in the reserve, randomly chosen from the Wolong Household Registration list for 2006. The sample frame included all households regardless of whether they had enrolled in GTGP. After 5 revisits, 11 households did not have an eligible interviewee and 5 households refused resulting in 305 respondents and a 95% response rate. Of these 305 households, only one did not participate in the GTGP and was removed from this study. Similarly high rates of participation in GTGP (> 85%) have been found in other places in China (Xu and Cao 2002, Ge et al. 2006, Tao et al. 2006). The elicited information includes household economic and demographic status, characteristics of enrolled GTGP land plots, and expected sustainable, annual fuelwood production from that land. Interviewees were asked if they plan to reconvert each of their GTGP land plots to crop production if the program ends in 2008 assuming that the prices of crop products will be the same as they were in 2005 and people will be allowed by the government to reconvert their enrolled plots if they want.

Among the households in our sample, 98 of them (32.2%) planned to reconvert at least some of their GTGP land plots to crop production after the GTGP ends and payment ceases. The number of land plots for reconversion (166) accounts for 22.6% of a total of 735. This low reconversion rate is not unique. Even lower planned reconversion rates (<20%) have been found in several other places in China (Bao et al. 2005, Liu 2005, Ge et al. 2006). Although the reconversion rate is low, the land plots chosen for reconversion are important to ecosystem services, such as the connectivity of panda habitat in Wolong, because they are often scattered among enrolled land plots.

2.2.2 Stated choice

Respondents who would reconvert all or some of their enrolled plots if the program ends were further questioned about their potential actions in the face of similar PES programs (e.g., extensions of the GTGP). Three contingent behavior questions were asked about their plans to re-enroll their land plots under different policy scenarios. Since actual behaviors in response to these scenarios cannot be observed, we asked respondents' intentions under these scenarios.

The proposed policy scenarios consisted of three attributes: conservation payment, program duration, and neighbors' behaviors. Each of these attributes had three levels. The amount of annual conservation payment ranged from 100 to 300 yuan/mu with an intermediate value of 200. After the first quarter of the survey, the high payment level was adjusted to 250 yuan/mu because almost all respondents would re-enroll all of their GTGP land plots under the annual payment of 300 yuan/mu, and changing the value to 250 yuan/mu allowed more variation in responses. The duration of proposed policy scenarios could be 3, 6, or 10 years. Neighbors were referred to as households who were located in the same group[1]. There were 26 groups within 6 villages within 2 townships in the reserve containing ≈1200 households, and each group contained from 14 to 89 households (Wolong Nature Reserve 2005). We defined households in the same group as neighbors because our respondents clearly know who are in the group, and households in the same group tend to have more interactions among each other, e.g., in collaborative planting and harvesting, which are important for social norms to be formed and sustained (Elster 1989, Coleman

1 In rural China, a group is a well-defined administrative unit within a village, and a village is an administrative unit within a township.

1990, Manski 2000). For the neighbors' behaviors, respondents were told that 25%, 50%, or 75% of households in the same group would reconvert part or all of their enrolled land plots. Therefore, there were 27 possible combinations of attribute levels.

In stated choice models, it is generally impractical and statistically inefficient to include all possible combinations of attribute levels within an experimental design (Louviere et al. 2000). Instead, a subset of the attribute combinations that maintains independent variation among the attributes is usually used in the choice questions. To understand the main effect of each scenario attribute on the program re-enrollment choices, we used a main effects design in which each of the three attribute arrays are orthogonal to one another (Hedayat et al. 1999). Each of the attribute combinations from the main effects plan then represents one of the "scenarios" presented in the stated choice question. In this study for each household before the interview, the scenarios were randomly drawn without replacement from the 9 scenarios from the main effects plan, and stated-choice methods (Louviere et al. 2000) were used to query people's re-enrollment intentions for GTGP land plots under different policy scenarios.

In the statistical analysis of the stated choice responses, both conservation payment and neighbors' behaviors entered as continuous variables (see econometric model below). This is common in stated choice models (Louviere et al. 2000) and allows model-based inferences of respondents' land use plans at attribute levels other than the design levels. For instance, given different levels of neighbors' reconverting rate, conservation program re-enrollment was evaluated across different levels of payment (0 ~ 300 yuan) where all other explanatory variables were set as their mean values as in Figure 2.1.

As suggested by two very influential theories of intention-behavior relationship, namely the theories of reasoned action and planned behavior (Fishbein and Ajzen 1975, Ajzen and Fishbein 1980, Ajzen 1985), intention is very often the strongest predictor of actual behavior (Madden et al. 1992, Schultz and Oskamp 1996, Terry and Hogg 1996). These theories specify three conditions that affect the magnitude of the relationship between intention and behavior: the degree of correspondence of specificity between the measure of intention and behavior; the degree of an individual's volitional control of carrying out the intention; and stability of intention during the time of measurement and performance of the behavior (Fishbein and Ajzen 1975, Madden et al. 1992). To improve the consistency between people's intentions and actual behaviors, our policy scenarios specify conservation payment and program duration, which are also attributes of the current GTGP as well as many other PES programs, and therefore are familiar to respondents. In addition, we selected household heads or their spouses as our interviewees because they are usually the decision makers for household affairs, and have the most volitional control over participation in PES programs.

2.2.3 Econometric model

We assume that farmers are willing to re-enroll their GTGP land plots in a renewed program if the utility of re-enrolling the plot is greater than the utility of the plot without re-enrollment. That is, $U_i^1 > U_i^0$, where U_i^1 and U_i^0 are the utilities of plot i being re-enrolled and not re-enrolled in the new program, respectively. The utility function U(.) is unobservable; however, there is a probability of re-enrolling $\Pr(Y_i = 1) = \Pr(U_i^1 > U_i^0)$ where Y_i=1 if planned to re-enroll and 0 otherwise, and a farmer's participation plan regarding the plot i, Y_i, can be observed.

Empirically, the program re-enrollment under different policy scenarios was modeled with a random-effects probit model (Wooldridge 2002):

$$\Pr(enroll_{ijk} = 1 \mid H_i, P_{ij}, S_{ik}, u_{ij}) = \Phi(H_i\alpha + P_{ij}\beta + S_{ik}\gamma + u_{ij}), \qquad (2.1)$$

where $\Pr(enroll_{ijk} = 1)$ is the probability of the ith household enrolling its jth GTGP land plot under the kth scenario; $\Phi(.)$ is the cumulative normal distribution; H_i represents household economic and demographic conditions as well as characteristics of the respondent associated with the ith household; P_{ij} represents the features of the jth land plot of the ith household; S_{ik} is the kth scenario that household i is exposed to; α, β, and γ are parameter vectors associated with household, plot, and policy scenario factors, respectively; and u_{ij} represents the unobserved random effects associated with the jth land plot of ith household. We did not find multicollinearity among independent variables. Since our goal was to obtain a relatively accurate estimation of the effects of independent variables, especially social norms, on the re-enrollment, the predictive power of the model is less important (Wooldridge 2003).

We used marginal effects to interpret the changes in the probability of re-enrollment in response to per unit change in explanatory variables. In cases where one is interested in the percentage changes in outcomes in responses to a percentage change in explanatory variables, elasticity should be estimated. In the probit model, the marginal effects of continuous variables are obtained from the formula (Greene 2003):

$$\frac{\partial \Pr(enroll = 1)}{\partial X} = \phi(X\beta)\beta, \qquad (2.2)$$

where X represents all model variables; $\phi(.)$ is the standard normal density function; and the derivative is calculated at the mean of the explanatory variables. The marginal effect for a dummy variable (d) is given by

$$\Pr(enroll = 1 \mid \bar{x}(d), d = 1) - \Pr(enroll = 1 \mid \bar{x}(d), d = 0), \qquad (2.3)$$

where $\bar{x}(d)$ represents the means of all other variables in the model.

2.3 RESULTS

2.3.1 Effects of social norms and conservation payment on re-enrollment

Both social norms and conservation payments had significant impacts on the respondents' intentions of re-enrolling their GTGP land plots in PES programs (Table 2.1). It was estimated that an additional 10% of neighbors' reconverting at least part of their GTGP land plots to agriculture reduced the respondents' intentions of re-enrollment by 6.4% on average. In other words, people's re-enrollment intentions can be affected by the re-enrollment decisions of their neighbors and tend to conform to the majority. With a decreasing proportion of neighbors' reconverting at least part of their GTGP land plots to agriculture, an individual's probability of program participation will increase. For instance, with an annual payment of 200 yuan/mu, 25% more land plots will be re-enrolled if the percentage of neighbors reconverting their GTGP land plots is changed from 75% to 25%.

The proposition of higher conservation payments increased the number of land plots intended for re-enrollment. Specifically, an additional yuan in the payment will increase the probability of re-enrolling in the PES program by 0.8%. Among the GTGP land plots that are likely to be reconverted to agriculture when the GTGP ends, more than half can be prevented from being reconverted under a PES program offering an annual payment of 200 yuan/mu. If the current GTGP can be renewed with the same payment (250 yuan/mu), more than 90% of GTGP land plots could be saved from reconversion. This finding is quite different from that in studies of the Conservation Reserve Program (CRP) in the United States, where maintaining enrolled land was much more expensive than the original cost (Cooper and Osborn 1998). Compared to the land set aside in the CRP, the GTGP land plots have high costs of reconversion due to reforestation in the land plots. Moreover, the GTGP land plots may provide additional ecosystem services, such as fuelwood production, to participants.

Intentions of re-enrollment were also influenced by the interactions between the conservation payment and neighbors' re-enrollment behavior (Figure 2.1). For instance, offering an annual payment of 200 yuan/mu with 75% of neighbors' reconverting at least part of their GTGP land plots had similar effects on the total re-enrollment as offering an annual payment of 158 yuan/mu with only 25% of neighbors' reconverting their GTGP land plots. Re-enrollment of 50% of land plots that will be reconverted when the GTGP ends would require an annual conservation payment of 184 yuan/mu or 142 yuan/mu if 75% or 25% of local residents were to reconvert at least part of their GTGP land, respectively. If the cost of program re-enrollment over multiple years and across all involved regions is considered, the differences in conservation cost under different social norms are substantial.

The impact of social norms on program re-enrollment was nonlinear across different levels of conservation payments. Social norms had the largest impact on the re-enrollment rate when the payment was intermediate, whereas the effects of social norms were smallest with the highest and lowest payments, where almost all or none of the respondents would participate (Figure 2.1).

2.3.2 Effects of program durations on re-enrollment

Program durations also had nonlinear effects on re-enrollment. As shown in Table 2.1, a 3-year program re-enrolled 23% fewer GTGP land plots than a 6-year program. However, re-enrollment for a 10-year program was not significantly different from a 6-year program (Table 2.1). Presumably, farmers made tradeoffs among stability, total payment, risks, and flexibility. Compared to short-term programs, longer-term programs provide more stable income and larger cumulative payment, but also bring more risks and less flexibility by limiting farmers' ability to adapt to changing conditions in markets of crop products.

2.3.3 Effects of household economic and demographic conditions

We found that sources of household income had different effects on program re-enrollment (Table 2.1). Farming income had a significant, negative effect on people's re-enrollment intentions. It was estimated that 1,000 more yuan of farming income reduced the probability of re-enrollment by 2.9%. However, income from off-farm employment outside of Wolong significantly increased the number of GTGP land plots to be re-enrolled in the PES program: 1,000 more yuan of income from employment outside of Wolong increased the probability of re-enrollment by 9.7%, whereas the incomes from off-farm employment within Wolong (tourism employment, temporary off-farm employment, and permanent employment) did not have such an effect. Although there are conflicts of time allocation between off-farm employment and farming, off-farm employment within Wolong is much more flexible in terms of time allocation compared to off-farm employment outside of Wolong, and therefore cause less conflicts with farming. Thus, not all off-farm income may increase the participation in PES programs, and different types of off-farm employment should be treated differently.

Households with more cropland tended to re-enroll their GTGP land plots in the PES program (Table 2.1) because GTGP land plots are usually marginal for growing crops, and people would not reconvert them to agriculture as long as they already have adequate land for farming. One extra mu of cropland increased the probability of re-enrollment by 13.8% (Table 2.1). In contrast to other studies (Cooper and Osborn 1998), livestock breeding did not affect people's re-enrollment intentions. Moreover, no effects of household size and total area of GTGP land plots on program re-enrollment were found (Table 2.1).

2.3.4 Effects of land plot features and respondent's characteristics

The respondents' perception of fuelwood that can be sustainably produced[2] by land plots had a positive effect on the program re-enrollment. Fuelwood is one of the most important energy sources for local people in Wolong (An et al. 2002). Since fuelwood collection will be allowed in the mature GTGP land, the prospect of more fuelwood production can increase the number of land plots to be re-enrolled. An expectation that the land plot will annually produce an additional 10 kg of fuelwood

[2] The amount of fuelwood that can be generated in land plots in the long run was estimated by respondents based on their past experiences of fuelwood collection.

in the long run increased the probability of re-enrolling the land plot by 1% (Table 2.1). Among households, the average distance from each household to its land plots had a negative effect on the program re-enrollment, probably because the average distance was correlated to some unmeasured variables, such as social status, of households. But within a household, the deviation of plot-household distance from the average distance (difference between each plot-household distance and the average distance of each household) was not significant in determining the GTGP land plots to be re-enrolled (Table 2.1). No effects of other plot features on the re-enrollment were found (Table 2.1).

For the characteristics of respondents, older people were more likely to re-enroll their GTGP land plots. One additional year of a respondent's age increased the probability of re-enrollment by 3.0% (Table 2.1). Since farming and reconverting GTGP land plots to agriculture are labor intensive, re-enrolling these land plots in the PES program would be a convenient way for older people to reduce labor demand (Nagubadi et al. 1996, Zbinden and Lee 2005). Respondents' gender also affected the program re-enrollment. Male respondents were 30.0% less likely to re-enroll their GTGP land plots than female respondents (Table 2.1). Combining gender effects with a respondent's age, on average a 50-year-old man had the same likelihood of re-enrollment as a 40-year-old woman. A respondent's education level was not found to affect program re-enrollment (Table 2.1).

2.4 DISCUSSION

Our findings suggested that the aggregate impacts of social norms at the neighborhood level on the cost of PES programs can be substantial. If most people in a community were to enroll their land in a conservation payment program, the extra cost for conserving an additional unit of land would be low due to social norms. Even in communities where most people would initially not participate in a PES program, social norms can be leveraged with increased conservation investments toward participation. Thus, incremental cost of conserving an additional unit of land can be reduced when social norms are leveraged.

A sustainable gain from PES programs can be achieved only if participants are willing to maintain conservation benefits, even after programs end (Uchida et al. 2005). As an alternative source of income to farming, off-farm employment through rural-to-urban labor migration not only lowers farmers' dependence on the enrolled land, but also reduces their ecological impacts (Liu et al. 2007a). Numerous off-farm employment opportunities have been generated by the transitional economy in urban areas of China (Yang 2000, Li and Zahniser 2002) and many other developing countries (Korinek et al. 2005). The trend of rural-to-urban migration is expected to continue over the next several decades (United Nations 2004). These labor and income trends provide a great opportunity for PES programs to lower costs and sustain conservation.

Economists have recognized that individuals' preferences over alternatives may depend on the actions of others (Manski 2000), suggesting that not only economic incentives but also social norms may be analyzed by means of utility theory

(Lindbeck 1997). Observed outcome data typically have limited power to distinguish the inference of social norms from other processes (Manski 2000). With the main-effects design of our stated-choice model, however, the inference of social norms can be relatively easily distinguished from the effects of other factors.

In conclusion, the results of our study suggest that the efficiency of conservation investments can be improved by integrating social norms at the neighborhood level with demographic trends, economic conditions, and biological values.

Table 2.1. Estimation of policy attributes and other characteristics and their marginal effects on the program re-enrollment.

	Independent Variables	Parameters	SE	Marginal Effects
Social norms and conservation payment	Neighbors' behavior	-1.662***	0.581	-0.636
	Conservation payment (yuan)	0.020***	0.003	0.008
Program durations	3-year duration (dummy, reference = 6 years)	-0.598**	0.277	-0.230
	10-year duration (dummy, reference = 6 years)	-0.270	0.281	-0.104
Household economic and demographic conditions	Farming income (1000 yuan)	-0.075*	0.042	-0.029
	Off-farm income (1000 yuan)			
	- Labor migration to outside of Wolong	0.253**	0.127	0.097
	- Tourism employment in Wolong	0.046	0.071	0.018
	- Temporary employment in Wolong	0.063	0.062	0.024
	- Permanent employment in Wolong	0.054	0.047	0.021
	Cropland after GTGP (mu)	0.361***	0.127	0.138
	Livestock (dummy)	0.406	0.520	0.157
	Household size	-0.127	0.176	-0.049
	Total land enrolled in GTGP (mu)	0.025	0.085	0.010
Land plot features	Area of land plot (mu)	-0.110	0.246	-0.042
	Fuelwood production (kg)	0.003*	0.002	0.001
	Average walking distance from each household to its land plots (minutes)	-0.038***	0.012	-0.015
	Deviation of plot-household distance from the average distance (minutes)	0.015	0.013	0.006
	Elevation (1000 m ASL.)	0.050	2.099	0.019
	Slope (degrees)	-0.038	0.024	-0.015
	Aspect (180 = north-facing; 0 = south-facing)	-0.007	0.006	-0.003
	Labor cost of reconversion (persons*days)	0.002	0.003	0.001
	Geographic location (dummy)	0.498	0.816	0.185
Respondent characteristics	Age (years)	0.077***	0.023	0.030
	Gender (reference = female)	-0.841*	0.474	-0.300
	Education (years)	-0.049	0.072	-0.019
Constant		-3.812	4.902	

Significance: * $p \leq 0.1$; ** $p \leq 0.05$; *** $p \leq 0.01$.

Observations = 498; Number of plots = 166; Log Likelihood = -219.209

Significant parameters for σ_μ = 1.836 (p < 0.01) and ρ = 0.771 (p < 0.01) suggest the random-effects model is appropriate, and the test statistic χ^2 =80.59 (p < 0.01) indicates the random-effects model is preferred to the model without random effects.

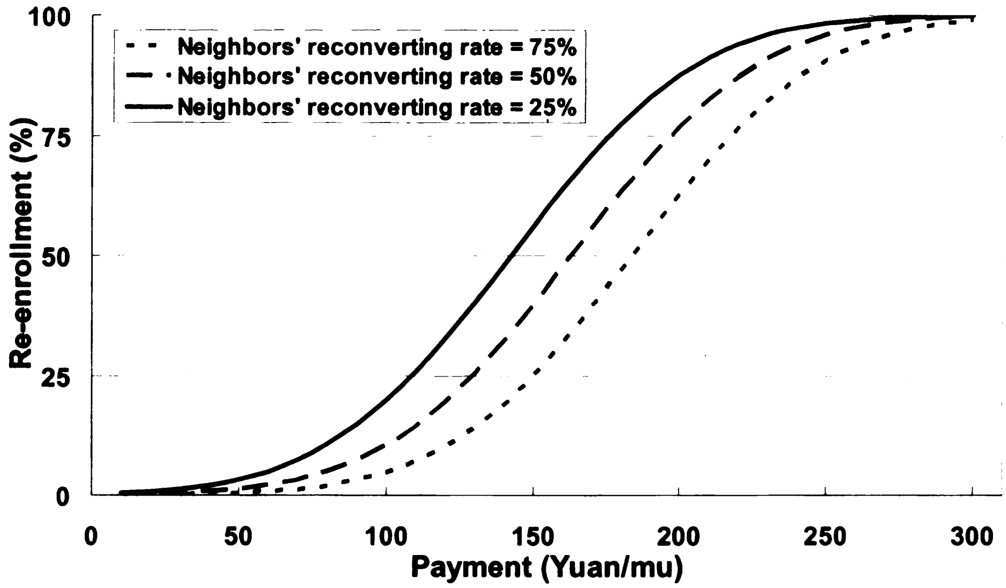

Figure 2.1. Estimated program re-enrollment under different levels of payment and neighbors' reconversion behavior.

CHAPTER 3

USING COST-EFFECTIVE TARGETING TO ENHANCE THE EFFICIENCY OF CONSERVATION INVESTMENT IN PAYMENTS FOR ECOSYSTEM SERVICES

In collaboration with

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ABSTRACT

Ecosystem services are being protected and restored worldwide through payments for ecosystem services. The efficiency of such investments depends on the design of payment scheme. Land features have been used to measure the environmental benefits of and cost for land enrollment in cost-effective targeting of land obtained through payments for ecosystem services. Household characteristics of program participants, however, may also be important in the targeting of land for enrollment. We used the characteristics of households participating in China's Grain-to-Green program and features of enrolled land to examine the targeting of land enrollment in that program in Wolong Nature Reserve. We compared levels of environmental benefits that can be obtained through cost-effective targeting of land enrollment for different types of benefits under different payment schemes. The efficiency of investments in a discriminative payment scheme (payments differ according to opportunity costs, i.e. landholders' costs of forgoing alternative uses of land) was substantially higher than in a flat payment scheme (same price paid to all participants). Both optimal targeting and suboptimal targeting of land enrollment for environmental benefits achieved substantially more environmental benefits than random selection of land for enrollment. Our results suggest that cost-effective targeting of land using discriminative conservation payments can substantially improve the efficiency of investments in the Grain-to-Green program and other payment for ecosystem services programs.

3.1 INTRODUCTION

Conservation programs in which landholders are paid to alter land management to achieve environmental benefits have been implemented in many countries (OECD 1997, Wunder 2008). These programs have reduced soil and wind erosion (Osborn et al. 1993), restored desirable attributes of ecosystems (Sierra and Russman 2006), and maintained habitat for native plants and animals (Johnson and Schwartz 1993, McMaster and Davis 2001). We refer to these desired changes or maintenance as environmental benefits. The efficiency of the investment in payment for such environmental benefits, often called payments for ecosystem services (PES), however, depends on the program's design.

To induce landholders to participate in PES programs, incentives should be greater than the cost of forgoing certain other uses of the land (i.e., opportunity costs). Landholder opportunity costs and the level of environmental benefits a parcel of land offers will vary. In practice, flat payments (all participants paid the same price) and discriminative payments (participants paid different prices according to opportunity costs) have been used in PES programs (Claassen et al. 2008, Pagiola 2008). At first glance, flat payments appear equitable because every participant is paid the same price. However, flat payments are not equitable when landholders bear different opportunity costs and their lands supply different levels of environmental benefits (Ferraro 2008). Additionally, discriminative payments where participants are paid their opportunity costs will cost less than flat payments. As such, society gains more environmental benefits for any given investment (Jack et al. 2008).

To maximize environmental benefits, PES programs must be implemented on land that provides the desired environmental benefits with the least cost which is referred to as cost-effective targeting or optimal targeting (Babcock et al. 1996). In a cost-effective targeting approach, a benefit-to-cost ratio (level of environmental benefits provided: cost) is used to rank plots of land from high to low. The lands with the highest benefit-to-cost ratio are enrolled in the PES program first so that a maximum amount of environmental benefits can be obtained with a fixed budget.

Lands enrolled in PES programs often supply multiple environmental benefits. Cost-effective targeting for one type of environmental benefit, however, usually does not maximize the provision of the other types of environmental benefits under a fixed budget unless the benefits are perfectly and positively correlated (Babcock et al. 1996). Therefore, the targeting approach that is optimal for a given environmental benefit is usually a suboptimal targeting approach for other types of environmental benefits (Babcock et al. 1996, Ferraro 2003). Nevertheless, where different types of environmental benefits are positively correlated, cost-effective targeting for achieving one environmental benefit will increase the level of other types of environmental benefits.

The environmental benefits provided by a particular parcel depend on the biological and physical features of the land and on the landholder's actions. In many cases, however, direct measurement of environmental benefits may be impossible or prohibitively expensive. Other researchers have used site-specific proxies of environmental benefits as measures of environmental benefits of land within PES programs. These proxies include a single biological or physical feature of land parcels

(Babcock et al. 1997, Siikamaki and Layton 2007) or combinations of biological and physical features (Babcock et al. 1997, Khanna et al. 2003, Ferraro 2004, Alix-Garcia et al. 2008).

Opportunity costs of landholders participating in PES programs are often difficult to measure because they are only known to landholders. However, landholder's opportunity costs are often correlated with the location and features of the land and with household characteristics (Cooper and Osborn 1998). Researchers have estimated the value of a parcel on the basis of its biological and physical features (Ferraro 2003, Khanna et al. 2003, Alix-Garcia et al. 2008) as proxies for the opportunity costs of landholders. Even though households are often the basic unit on which land-use decisions are based (Liu et al. 2003a), household characteristics of landholders usually have not been included in determination of opportunity costs (Naidoo et al. 2006, Siikamaki and Layton 2007). Despite these measurement difficulties, targeting in PES programs can substantially improve the efficiency of investments, especially when the level of environmental benefits and the costs to obtain the benefits are heterogeneous across the parcels within a landscape (Osborn et al. 1993, Babcock et al. 1996, 1997, Chan et al. 2006).

In actual implementation of PES programs, it may not be feasible to collect the information on households and land parcels needed to determine opportunity costs for cost-effectively targeting of lands to enroll. Another approach to enrolling lands in PES programs is to use competitive auctions in which potential enrollees submit bids (the payment they require) to provide environmental benefits. The cost-revelation mechanism in most competitive bidding processes makes auctions a powerful tool for inducing potential participants of PES programs to submit bids equal to their opportunity costs (Latacz-Lohmann and Van der Hamsvoort 1997).

China is implementing several large-scale conservation programs (Liu et al. 2008). Among these is the Grain-to-Green program (GTGP), which was implemented in 1999 and is the largest PES program in the developing world. Participating farmers receive payments in grain or cash for a maximum of 8 years to convert cropland to forest or grassland. Because the main objective of GTGP is to reduce soil erosion by increasing natural land cover (forest and grassland), the slope of enrolled land should be above 15° in northwestern China and above 25° elsewhere. Although croplands with slopes above the thresholds receive priority for enrollment, some croplands with slopes lower than the thresholds have been enrolled (Uchida et al. 2005). By the end of 2006, GTGP had converted about 9 million ha of cropland into forest and grassland (Liu et al. 2008). (In the United States, about 14.5 million ha of cropland are enrolled in a similar program, the Conservation Reserve Program (Claassen et al. 2008) In addition to its main objective of restoring natural vegetation cover, GTGP aims to generate other environmental benefits, such as restoration of habitat for certain animals and plants (Zuo 2002b).

The GTGP has only 2 payment levels nationwide which operate as flat payments within each region. On an annual basis payments are 2250 kg of grain or 3450 yuan per ha of enrolled cropland in the upper reaches of the Yangtze River basin and 1500 kg of grain or 2400 yuan in the middle-upper reaches of the Yellow River basin. The different regional payment levels are used in part to account for the regional differences in opportunity costs of landholders because land in the upper reaches of the Yangtze River basin is usually more productive than in the middle-upper reaches of the Yellow River basin (Uchida et al. 2005). The payments for most participating farmers exceed cultivation income from the enrolled land (Uchida et al. 2009), which indicates similar environmental benefits may be obtained at lower cost. By the end of 2005, more than 90 billion yuan had been invested in GTGP (Liu et al. 2008). When contracts started expiring in 2008, they were extended for up to 8 years. In the future the program's budget is likely to be reduced (Liu et al. 2008). Given its large scale and heterogeneities in opportunity costs of landholders and environmental benefits, the cost-effectiveness of GTGP payments may be improved greatly if payments are made to landholders whose lands can provide environmental benefits at lowest cost (i.e., cost-effective targeting of lands to be enrolled). We examined the GTGP in China's Wolong Nature Reserve to determine the efficiency of investments made through cost-effective targeting using flat and discriminative payments to landholders. We used features of specific parcels as proxies of environmental benefits and physical features of the parcels and household characteristics of landholders to estimate the opportunity costs of participating in GTGP. The results of our study can be used to maintain the environmental benefits from GTGP after the expiration of current contracts with a reduced budget.

3.2 METHODS

3.2.1 Modeling strategy

To study cost-effectiveness of alternative GTGP targeting and payment schemes, we modeled enrollment in and environmental benefits from GTGP in Wolong Nature Reserve. We determined the locations, environmental benefits, and opportunity costs for all GTGP plots in the reserve. Because we did not know GTGP plot locations of all households, we distributed all GTGP plots across the landscape through stochastic simulations and then calculated environmental benefits provided by these plots. We then modeled the conversion to croplands of GTGP plots that were not re-enrolled in the program after cessation of payments and re-enrollment of plots in the program. We used these models to identify the enrollment probabilities and opportunity costs for each GTGP plot. We modeled the environmental benefits provided by the enrolled GTGP plots under different conservation budgets and compared cost-effectiveness among the different targeting approaches and payment schemes.

3.2.2 Household survey

We interviewed heads of households in Wolong Nature Reserve in the summer of 2006. We used the government's household registration list of 2006 to randomly select 321 of the 1200 households for interviews. Of those 321, 304 (95%) completed an interview. For each plot enrolled in GTGP, we collected information on the landholder's land-use plans after expiration of their GTGP contact. These plans were used to estimate the probability a household planned to reconvert their GTGP plot after the current program ends, *P(reconvert)*. Surveyed landholders planned to convert 166 (22.6%) of their 735 GTGP plots to crop production after GTGP payments ceased (Chen et al. 2009a).

For those respondents that planned to reconvert their GTGP plots, stated choice methods (Louviere et al. 2000) were used to elicit whether they would re-enroll in a new round of GTGP under various payment amounts, i.e.,

$P(reenroll_j \mid pay > 0, reconvert)$. The proposed annual conservation payment had three levels: 1,500, 3,000, and 4,500 Yuan per ha. After the first quarter of the survey, the highest level of payment was adjusted to 3,750 Yuan per ha because almost all respondents would re-enroll all of their GTGP plots under an annual payment of 4,500 Yuan per ha, and changing the value to 3,750 Yuan per ha allowed more variation in responses. Varying payment prices across scenarios and respondents allowed us to statistically model re-enrollment as a function of payments, thus to identify opportunity costs of re-enrollment.

3.2.3 GTGP land identification

For all households in the reserve, we obtained information on characteristics such as household size and age and gender of the household head from the local government's 2006 household registration list. The geographic location of each household in the reserve was recorded in 2006 with global positioning system (GPS) receivers. Government data for the reserve showed that 2470 plots (total of 367.5 ha) belonging to 969 households were enrolled in GTGP in 2003.

Although information on the number of plots each household had enrolled in the program and the area of each plot was available, information on the geographic location of the plots was not available. Since we required information on the locations of all plots that were enrolled in the GTGP, we developed a map of the probability that each grid cell (i.e., pixel) is under the GTGP. The 304 surveyed households enrolled a total of 735 plots comprising a total of 110.4 ha. The locations of these 735 plots were measured using a GPS receiver. To map the probability that each grid cell is under the GTGP, we used a fuzzy classification algorithm based on the principle of maximum entropy (Jaynes 1957). The algorithm was applied to multi-spectral and topographic data in grid format using the software MaxENT (Phillips et al. 2006). Multispectral data consisted of two Landsat Thematic Mapper (TM) images (28.5m x 28.5m / pixel) acquired on April 19 and September 18, 2007. Topographic data (with the same pixel resolution as the Landsat TM imagery) consisted of elevation, slope and aspect derived from a digital elevation model generated for the study area from topographic maps (Liu et al. 2001). We randomly selected two-thirds of the geographic locations of the 735 GTGP plots that we measured to calibrate the fuzzy classification algorithm, and one-third to validate the output map. Although the area of some GTGP plots is smaller than the area comprised by a Landsat TM pixel, if at least one GTGP plot fell within a pixel, the entire pixel was considered as a GTGP plot, and used for model calibration and validation. This constitutes an approximation since not necessarily 100% of a pixel is under the GTGP, however it is a common procedure in many pixel-based imagery classification methods (Lu and Weng 2007). We then resampled the resolution of the GTGP probability map to 10 meters using nearest-neighbor interpolation so that each GTGP plot occupied at least one pixel. Since all of these 735 plots were located within 6 km of their corresponding households, the probability map (Figure 3.1) was developed in a 6 km buffer around all household locations.

The GTGP probability map was validated by means of a receiver operating characteristic (ROC) curve (Hanley and Mcneil 1982). The ROC curve is a plot of the sensitivity values (i.e., true positive fraction) vs. their equivalent 1-specificity values (i.e., false positive fraction) for all possible probability thresholds. The area under the

ROC curve (AUC) is a measure of model accuracy, with AUC values ranging from 0 to 1, where a score of 1 indicates perfect discrimination, a score of 0.5 implies a prediction that is not better than random, and lower than 0.5 implies a worse than random prediction. We used the validation data set (one-third of the 735 GTGP plots that we measured) together with 10,000 randomly selected pixels (Wiley et al. 2003, Phillips et al. 2006) for deriving the AUC value. The GTGP probability map exhibited high accuracy (AUC = 0.98).

We then stochastically distributed all 2470 GTGP plots across the landscape on the basis of the GTGP probability map and the probability distribution of distances between the 735 GTGP plots and their corresponding households. For each GTGP plot, we first randomly chose its distance to its corresponding household based on the probability distribution of the distances between the 735 GTGP plots and their corresponding households. We then randomly chose a pixel as the central pixel of the GTGP plot from all the pixels on the GTGP probability map that are at the specified distance from the household based on these pixels' probability of being GTGP land. Finally, the neighboring pixels, with a positive probability of being GTGP land, of the central pixel were treated as part of the GTGP plot until the area of the GTGP plot was reached. The simulation was conducted using Java programming language (JDK 1.4.2, Sun Microsystems).

3.2.4 Quantification of environmental benefits

To examine the different targeting approaches and payment schemes for GTGP, we constructed three possible indices of environmental benefits. Because slope was the only available measure of the reduction of soil erosion through GTGP

(Uchida et al. 2005), we used plot-specific slope (measured by the mean slope of pixels in the plot) as a proxy for the environmental benefit of reduction in soil erosion. The soil-benefit index is the square of the standardized (Ferraro 2004) slope of the plots:

$$\text{soil benefit index}_i = \left(\frac{\text{slope}_i - \text{slope}_{\min}}{\text{slope}_{\max} - \text{slope}_{\min}}\right)^2, \tag{3.1}$$

where slope_i is the slope of a GTGP plot and $\text{slope}_{\min}$ and $\text{slope}_{\max}$ are the minimum and maximum slopes among all GTGP plots, respectively. This index measures the relative steepness of a plot relative to the minimum and maximum slopes among all GTGP plots in the reserve. The higher the soil-benefit index of the ith plot ($\text{soil benefit index}_i$), the greater the probability the ith plot will have less soil erosion if enrolled in the PES scheme than if used to grow crops. Because land with steeper slopes was given priority for enrollment in GTGP, we used the square of the standardized slope to place more weight on plots with steeper slopes. It should be pointed out that slope-based proxies of soil benefit do not represent all the factors, such as rainfall, soil type, and ground cover, which determine soil benefit. As these types of information become available, more robust measures of soil benefit, such as the Revised Universal Soil Loss Equation (RUSLE) (U.S. Department of Agriculture 2001), may be used.

Besides reducing soil erosion, GTGP also aims to restore habitat for many plant and animal species. Distance to patches of habitat that existed prior to PES programs has been used as a measure of habitat quality (Babcock et al. 1996). However, distance-based proxies of habitat quality do not represent all the factors important in determining habitat quality. We used the distance between GTGP plots and the nearest patch of natural forest as a measure of the habitat quality of the GTGP plot. Using protocols described in (Viña et al. 2007), we determined the distribution of natural forest (Figure 1.1) in the reserve by classifying remotely sensed imagery acquired on 18 September 2007. The habitat-benefit index is

$$\text{habitat benefit index}_i = \left(1 - \frac{\text{dist}_i - \text{dist}_{\min}}{\text{dist}_{\max} - \text{dist}_{\min}}\right)^2, \tag{3.2}$$

where dist_i is the distance between a GTGP plot and the nearest natural forest patch, $\text{dist}_{\min}$ and $\text{dist}_{\max}$ are the minimum and maximum distances to the nearest natural forest patches among all GTGP plots, respectively. Here we used a subtraction from unity so that a higher index value would correspond to a smaller distance to the nearest forest patch. Therefore, the higher the habitat-benefit index of the *i*th plot ($\text{habitat benefit index}_i$), the higher the habitat quality the *i*th plot is presumed to have for certain animals and plants. As with the soil-benefit index, we used the square of the standardized distance to place more weight on those plots that were closer to patches of natural forest.

We measured the amount of each type of environmental benefit of a plot by multiplying the benefit index by the area of the plot. For comparison purposes, we also measured the amount of land area enrolled in the PES program, defined as land benefit, when we examined the effectiveness of the different approaches to targeting environmental benefits and the different payment schemes.

3.2.5 Opportunity-cost estimation

Given that GTGP plots were still under contract when our data were collected, we used landholders' plans for their GTGP plots after their contracts expired to model the probability of land holder re-enrollment in GTGP. For those plots for which there were no plans to convert the land to crops after the contract expired, we assumed the plots would be re-enrolled under any positive payment for participation. For GTGP plots landholders planned to convert, there was a probability that the plot would be re-enrolled if any positive payments were offered. Thus, the probability of a GTGP plot being re-enrolled is

$$\begin{aligned} P(\text{re-enroll}_j) = 1 - P(\text{convert}_j) + \\ P(\text{convert}_j) * P(\text{re-enroll}_j \mid \text{pay} > 0, \text{convert}) \end{aligned}, \tag{3.3}$$

where $P(\text{convert}_j)$ is the probability of the *j*th GTGP plot being converted to crop production after contract expiration, $1 - P(\text{convert}_j)$ is the probability the *j*th GTGP plot will not be converted to crop production after contract expiration (and thus the plot will be re-enrolled at any positive payment), and $P(\text{re-enroll}_j \mid \text{pay} > 0, \text{convert})$ is the probability of re-enrolling the *j*th GTGP plot under a new payment program for plots that will be converted to crop production after contract expiration, which must then be weighted by the probability that the plot will be converted, $P(\text{convert}_j)$.

In logistic regression models, we used proposed conservation payments, features of GTGP plots, and household characteristics to explain the probability a GTGP plot will be re-enrolled (Eq. 3.3). We corrected for dependencies among plots of the same landholder and among responses to different proposed alternative payments for the same plot with Huber's variance correction (Wooldridge 2002). We did not find multicollinearity among independent variables. We applied these models to all GTGP plots in the reserve and calculated the probability of each GTGP plot being re-enrolled ($P(\text{re-enroll}_j)$ in Eq. 3.3). We determined the per hectare opportunity cost of each plot with a Bernoulli trial, which determined re-enrollment of plots as a function of the payment. The rate parameter of the Bernoulli distribution was, $P(\text{re-enroll}_j)$, and we estimated it for different payment amounts (Cooper and Osborn 1998). The per hectare opportunity cost of a plot was the payment level at or above which the plot would be enrolled. The opportunity cost of a plot was the per hectare opportunity cost of the plot multiplied by its area.

3.2.6 Environmental benefits targeting approaches

For each of the three types of environmental benefits, we examined the amount of that environmental benefit that can be obtained with cost-effective targeting of the lands to enroll in the PES program. We also illustrated, how much of each type of environmental benefit would be obtained had one of the other two types of benefits been the target of the PES program (i.e., sub-optimal targeting). We examined targeting using both flat payment and discriminative payment schemes. We conducted the initial analysis only on those GTGP plots that would be converted to crop production after contract expiration. Under the discriminative payment scheme, we determined the cost-effective enrollment of plots for each type of environmental benefit by ranking all GTGP plots from high to low according to the benefit that could be obtained for each unit of cost (i.e., ratio of benefit to cost) and enrolling plots with the highest benefit-to-cost ratio first. For the land-benefit maximization approach, where the goal is to maximize the area of land enrolled, we based GTGP plot enrollment on per hectare cost; thus, less expensive GTGP plots had enrollment priority. In addition to determining cost-effective targeting for each type of environmental benefit, we also calculated the amount of each environmental benefit obtained and the amount of land enrolled in GTGP under suboptimal targeting (i.e. when plot benefit-to-cost ratios are ranked on the basis of the non-targeted environmental benefits). For instance, maximizing the amount of land enrolled is the optimal approach for land acquisition, but it is usually suboptimal for acquiring either of the other environmental benefits. Maximization of soil benefits, however, is the optimal approach to achieve soil benefits, but it is suboptimal for improving habitat quality for some species and for land acquisition.

To understand the relation between each environmental benefit and expenditure, we calculated the total level of an environmental benefit that can be obtained within a budget that varied from zero to the cost of obtaining all the environmental benefits possible. Because our spatial distribution of GTGP plots and enrollment decision were stochastic processes, we calculated the mean values of environmental benefits from 300 simulations for each targeting approach to facilitate relatively robust relations between environmental benefits and expenditure. Results from a total of 400 simulations are almost identical to those from 300 simulations. We also drew a 45° line (Babcock et al. 1996) in each of the benefit-budget planes to show the amount of environmental benefit that could be obtained through random selection of plots constrained within a particular budget.

In addition to the discriminative payment scheme, we explored the environmental benefits obtained through a flat payment scheme. Under the flat payment scheme, all plots with per hectare opportunity costs less than or equal to the per hectare payment were enrolled. When all landholders are paid the same flat price for their plots, each increase in the number of plots enrolled requires that a higher per hectare payment be made to all plots, not just to the plots with higher opportunity costs. Thus, all plots that would have enrolled at a lower payment level (because their opportunity costs were lower) receive a surplus equal to the difference between their opportunity costs and the amount of the flat payment. The magnitude of this surplus defines the difference in costs between discriminative and flat payment schemes.

3.3 RESULTS

3.3.1 Effects of household characteristics and plot features

Household size had significant positive effects on the probability of conversion of a GTGP plot to cropland (Table 3.1). The more land the household had enrolled in GTGP, the less likely the household planned to convert any of the plots to agriculture. In addition, households in Gengda were less likely to have plans to convert plots than households in Wolong.

The higher the payment the more likely landholders were to participate in GTGP (Table 3.2). Households with more members were less likely to re-enroll their plots. Probability of re-enrolling increased as age of household head and area of cropland increased. The distance between plots and the household reduced the probability of re-enrollment, perhaps because distance was correlated with some unmeasured variables such as the household's social status.

3.3.2 Cost-effective targeting of land for environmental benefits

Our simulations showed that about 78% of GTGP land in the reserve would not be converted to agricultural uses even after the expiration of contracts. We conducted our analyses of re-enrollment of plots under new payments by including all plots and by including only those plots that would be converted to agriculture when contracts expired; in the figures that follow we show the results for the latter (Figure 3.2 – 3.5). The approach that optimized soil benefits (i.e. cost-effective targeting for soil benefits) obtained 82% of the soil benefits when the budget for payments was 100,000 yuan and 97% of the benefits when the budget was 200,000 yuan (Figure 3.2). Cost-effectively targeting for habitat benefits obtained 81% of the habitat benefits (Figure 3.3) when the budget for payments was 100,000 yuan, whereas cost-effectively targeting for land benefits obtained 75% of the land benefits (Figure 3.4) when the budget was 100,000 yuan.

Even though cost-effective targeting achieved more of the targeted environmental benefit for any budget amount than when suboptimal targeting was used, suboptimal approaches were far superior to random selection of plots. In all cases, differences in the amount of environmental benefits obtained between optimal and suboptimal targeting approaches were much smaller than differences between any of the targeting approaches and random selection of plots. When the budget for payments was 100,000 yuan (Figure 3.2), cost-effectively targeting for soil benefits obtained 82% of the soil benefits compared to 76% and 69% of the soil benefit from the two suboptimal approaches, but only 29% of the soil benefit was obtained when plots were randomly selected for enrollment (45° line).

The amount of environmental benefits obtained with discriminative payments and flat payments were quite different. It cost 92,000 yuan with discriminative payments to obtain 80% of soil benefits (Figure 3.2). To obtain the same amount of soil benefit with flat payments (Figure 3.5), it cost 298,000 yuan. The difference between the cost of discriminative payments and the cost of flat payments increased as the percentage of environmental benefits increased. For instance, in terms of land acquisition, to obtain 30%, 60%, or 90% of the land with flat payments would cost 29,000, 128,000, and 585,000 yuan, which is about 1.7, 2.1, and 3.4 times the cost of discriminate payments, respectively.

These differences demonstrate how efficiency of investments can be improved by switching from the most cost-effective flat payment approach to the most cost-effective discriminative payment approach. Results presented in the graphs illustrate the effectiveness of targeting specific levels of benefit and assume that no payments would be made to landholders who did not plan to convert lands to crop production upon contract expiration. As such, even flat payments are to a small degree discriminative payments. When we included in the payment scheme the GTGP plots that would not be converted to crop production after contacts expired, the efficiency of the payments improved by more than 10 times when we switched from flat payments to discriminative payments.

3.4 DISCUSSION

Substantially greater environmental benefits were obtained when lands were optimally or suboptimally targeted for enrollment than when enrollment of land was random. When suboptimal targeting approaches are used in PES schemes, the efficiency of the program depends on correlations among the types of environmental benefits (Babcock et al. 1997). When different environmental benefits of plots are highly and positively correlated, as in our case, similar amounts of environmental benefits can be obtained with suboptimal targeting as can be obtained with cost-effective targeting. More generally though, targeting the desired environmental benefit can be critical to achieving conservation objectives if the environmental benefits of plots are not highly and positively correlated.

The differences in cost-effectiveness between the payment schemes was substantially larger than the differences among environmental benefit targets. In all cases, discriminative payments were more efficient (up to 10 times) than flat payments. The reason for the difference is that flat payments pay all enrollees the same price regardless of opportunity costs.

Household characteristics were also significant determinants of opportunity costs of landholders participating in GTGP. For instance, a plot that has little agricultural value for a household with a small labor supply can be much more valuable for a household with a larger labor supply. In addition, we found substantial regional differences in landholders' willingness to continue participating in GTGP. One of the main differences between the 2 townships in our study was that Gengda was closer to more urbanized regions outside the reserve. Thus, PES programs are more likely to cost-effectively achieve their objectives if household characteristics and regional differences, as well as biological and physical features, are incorporated in the planning of PES programs, especially in areas without robust land markets. Other household characteristics (e.g., off-farm income) were also significant determinants of opportunity costs (Chen et al. 2009b), but were not included in this study because such information was not available for all households in the reserve.

Opportunity costs of landholders are typically private information that is not available to the public, which results in an information gap between landholders and conservation practitioners (Ferraro 2008). Competitive auctions can reduce this information gap substantially (Latacz-Lohmann and Van der Hamsvoort 1997). Moreover, competitive auctions have been applied successfully in some PES programs and have improved the efficiency of conservation investments (Kirwan et al. 2005, Claassen et al. 2008). Cost-effective targeting for environmental benefits coupled with competitive auctions could greatly improve the efficiency of investments in PES programs, especially in programs, such as GTGP, that are relatively large and have substantial heterogeneities in opportunity costs and environmental benefits. Competitive auctions and cost-effective targeting may increase transaction costs of PES programs, but our results suggest that the improved efficiency from cost-effective targeting will far outweigh likely increases in transaction costs in GTGP. Although cost-effective targeting using competitive auctions will likely improve the efficiency of conservation investments in PES, it is important to maintain low transactional costs. The growing demand for conservation resources globally (Ferraro 2008, Jack et al. 2008) makes it increasingly important to improve the efficiency of investments in PES and other conservation programs.

Table 3.1. Pooled logit estimation of conversion of Grain-to-Green program plots to agriculture after contract expiration.[a]

Independent variables	Description	Parameters (robust SE)	Marginal effects
Household size	no. of people in the household	0.250* (0.103)	0.039
Cropland	cropland of the household (ha)	-0.963 (1.022)	-0.151
GTGP land	land enrolled in GTGP (ha)	-1.734** (0.633)	-0.273
Age of household head	years	-0.003 (0.012)	-0.001
Gender of household head	1,female; 0,male	0.400 (0.376)	0.069
Township	1, Gengda township; 0, Wolong township	-1.182* (0.515)	-0.199
Area	ha	0.015 (1.018)	0.002
Slope	degree	-0.004 (0.016)	-0.001
Elevation	100 m (asl)	-0.033 (0.104)	-0.005
Distance	100 m	-0.050 (0.030)	-0.008
Constant		0.396 (2.346)	
χ^2		44.41***	

[a] $P(\text{convert}_j)$ in Eq. 3.3; number of plots 735.

Significance: * $p \leq 0.05$; ** $p \leq 0.01$; *** $p \leq 0.001$.

Table 3.2. Pooled logit estimation of re-enrollment of Grain-to-Green program plots after expiration of current contract.[a]

Independent variables	**Parameters (robust SE)**	**Marginal effects**
Ln(payment in yuan)	1.816*** (0.300)	0.453
Household size	-0.372** (0.141)	-0.093
Cropland	3.668** (1.169)	0.914
GTGP land	0.340 (0.839)	0.085
Age of household head	0.034** (0.013)	0.008
Gender of household head	-0.330 (0.469)	-0.082
Township	0.102 (0.521)	0.025
Area	0.994 (1.203)	0.248
Slope	0.016 (0.020)	0.004
Elevation	-0.003 (0.134)	-0.001
Distance	-0.096* (0.041)	-0.024
Constant	-9.985*** (3.071)	
χ^2	59.83***	

[a] $P(\text{re-enroll}_j \mid \text{pay} > 0, \text{reconvert})$ in Eq. 3.3; observations 498; number of plots 166.

Significance: * $p \leq 0.05$; ** $p \leq 0.01$; *** $p \leq 0.001$.

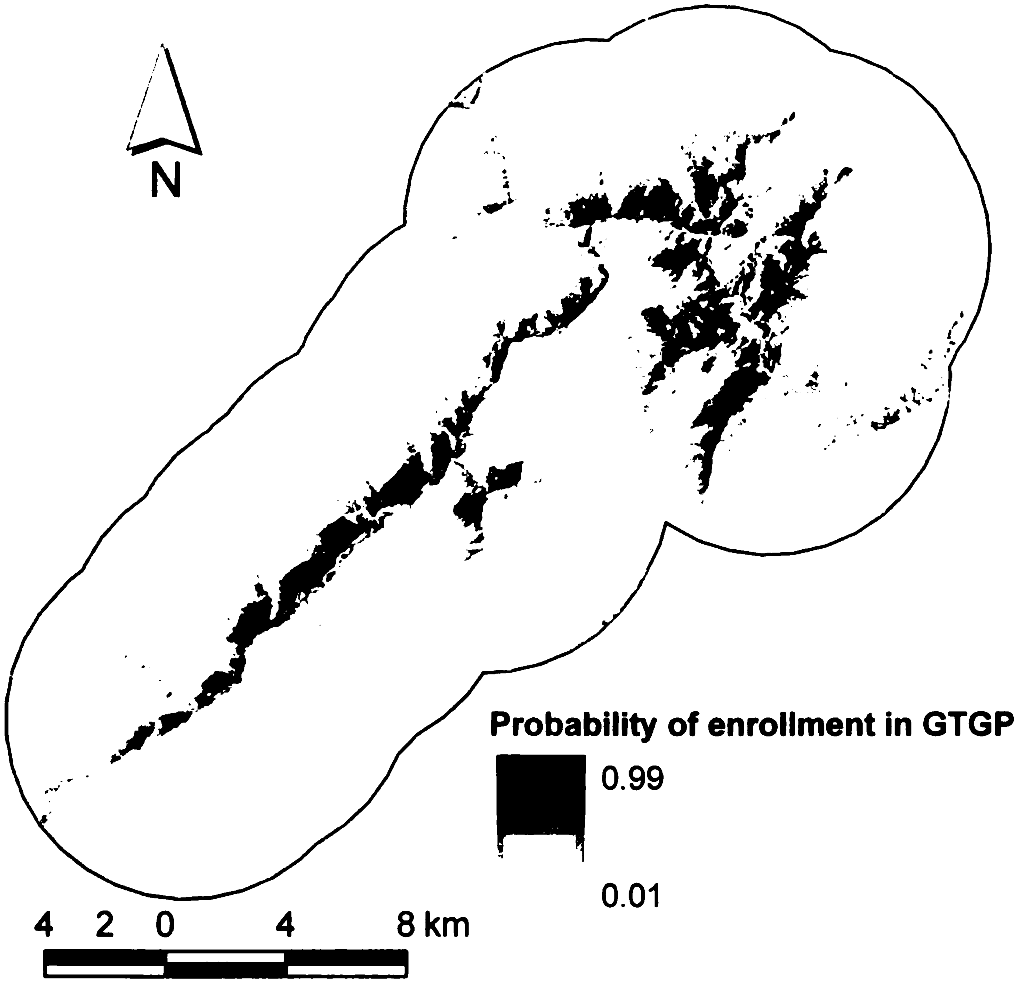

Figure 3.1. Probability of land being enrolled in the Grain-to-Green program.

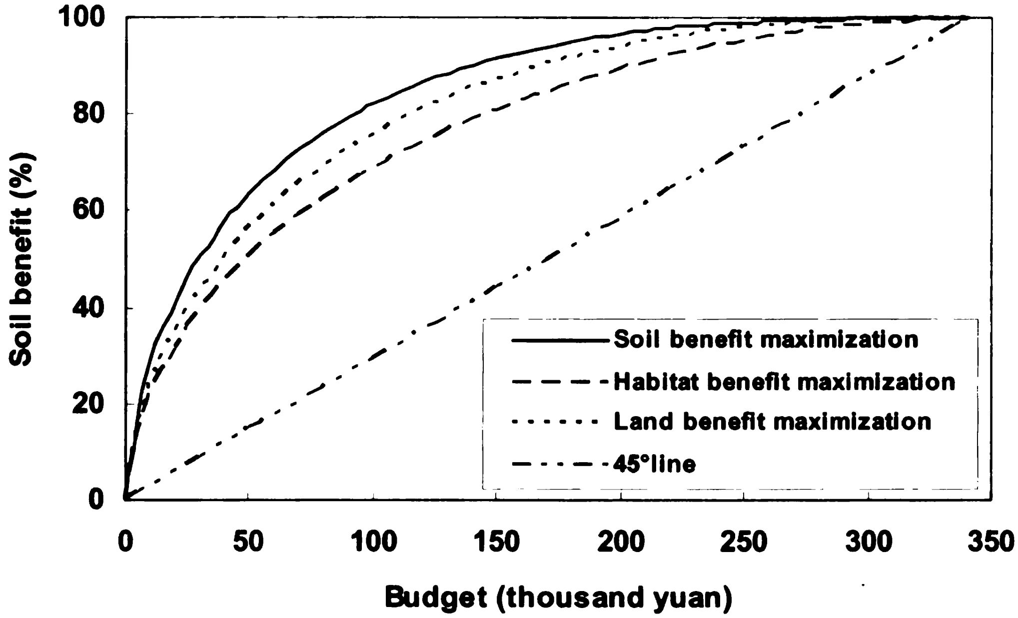

Figure 3.2. Percentage of soil benefit obtained with different benefit targets and budgets using a discriminative payment scheme.

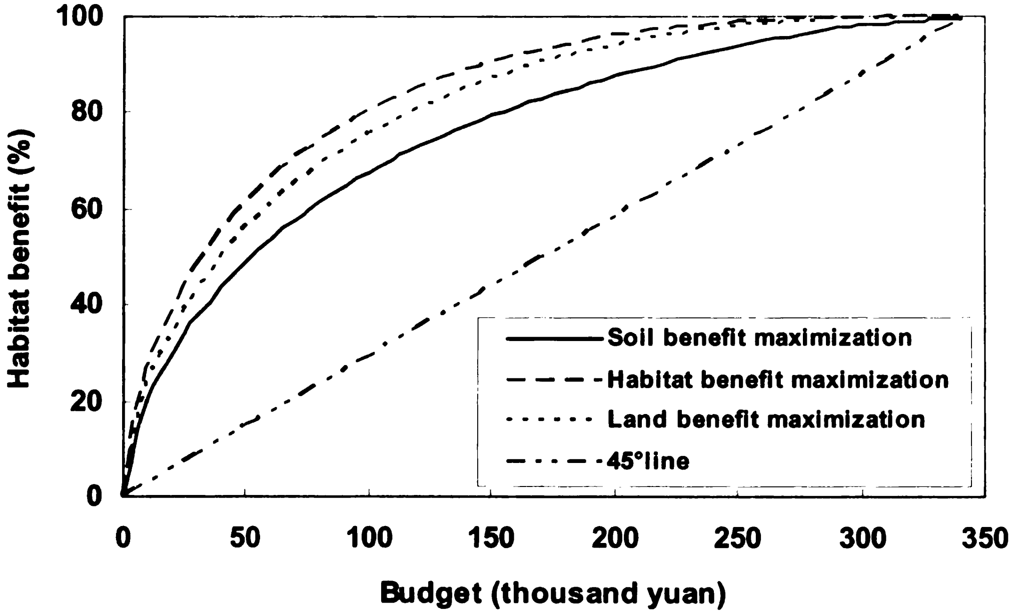

Figure 3.3. Percentage of habitat benefit obtained with different benefit targets and budgets using a discriminative payment scheme.

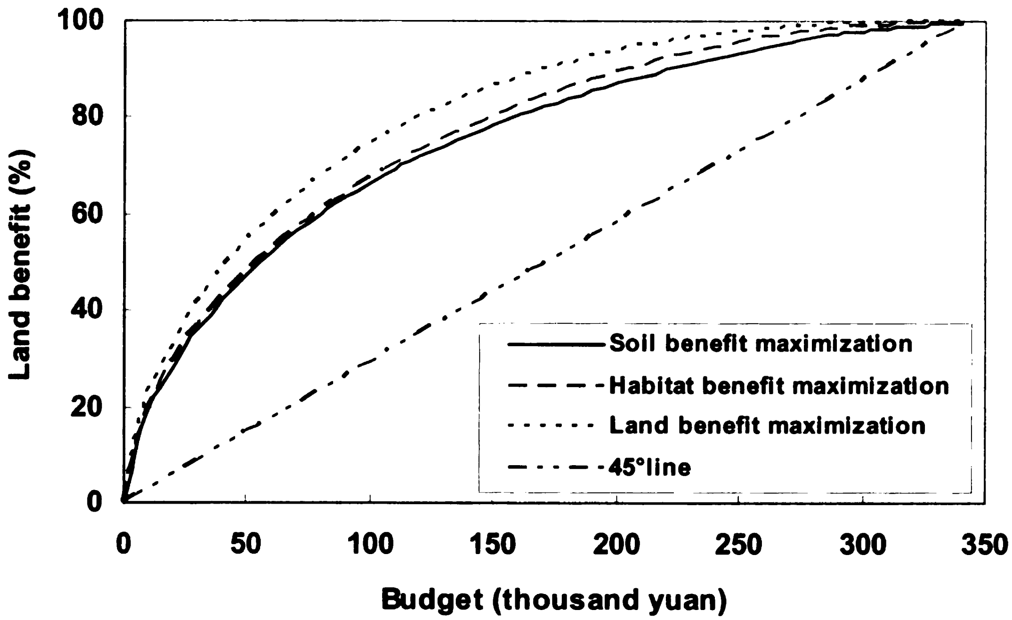

Figure 3.4. Percentage of land benefit obtained with different benefit targets and budgets using a discriminative payment scheme.

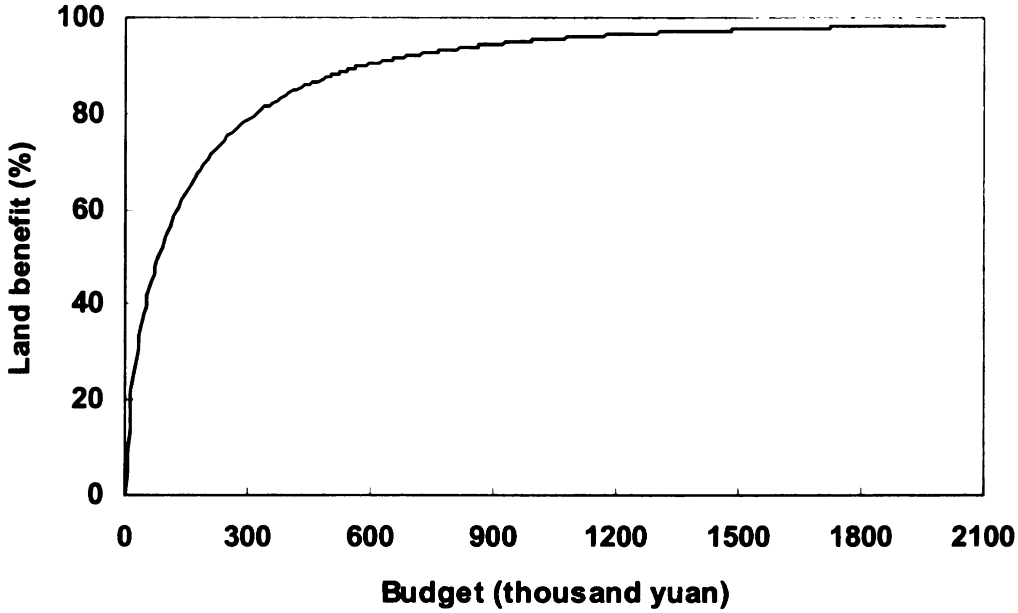

Figure 3.5. Percentage of land benefit obtained with different budgets using a flat payment scheme. Only the land-benefit (amount of land area enrolled in the PES program) curve is shown because the curves for soil benefits and habitat benefits are almost identical to the land-benefit curve.

CHAPTER 4

WEAK TIES, LABOR MIGRATION, AND ENVIRONMENTAL IMPACTS: TOWARDS A SOCIOLOGY OF SUSTAINABILITY

In collaboration with

Kenneth A. Frank, Thomas Dietz, Guangming He, and Jianguo Liu

ABSTRACT

Debate about the substitutability of manufactured, natural, human and social capital is at the heart of sustainability theory. Sociology can contribute to this debate by examining the processes and mechanisms by which one form of capital is substituted for another. We examine the substitution among different forms of capitals at China's Wolong Nature Reserve where the consumption of an important aspect of natural capital, fuelwood, has serious consequences for the local environment. We found that social capital in the form of weak social ties to people in urban settings had significant impacts on rural-urban labor migration. Following the chain of capital substitutions, labor migration then significantly affected income, which in turn affected fuelwood consumption. Quantification of the validity of inferences suggests the inferences are robust with respect to concerns about omitted confounding variables.

4.1 INTRODUCTION

At least since the Bruntland Commission published its historical report "Our Common Future" (World Commission on Environment and Development 1987), international policy has been concerned with the practices that affect sustainability (see also International Union for the Conservation of Nature 1980, U.S. National Research Council 1999, Rockwood et al. 2008). Sustainability also has spurred a vibrant literature in resource, environmental, development and ecological economics with some contributions from political science and philosophy (Becker and Ostrom 1995, Kates et al. 2001, Clark and Dickson 2003, Norton 2005, Henry 2009). However, aside from some critiques of the term as used in environmental politics

(Blühdorn 2007, Blühdorn and Welsh 2007), we cannot point to a sociological approach to sustainability. The challenge for sociology is to develop an approach to sustainability that moves past the focus on human action alone as in the human exceptionalism paradigm (Catton and Dunlap 1978, Dunlap 1994) to embrace the study of coupled human and natural systems (Liu et al. 2007a, Liu et al. 2007b).

The debate in sustainability theory over the degree to which "natural capital", defined as the goods and services humans derived from ecosystems (Costanza et al. 1997, Daily et al. 2000), can be replaced by "manufactured capital,[3]" in the form of increased affluence provides an entry point for sociology. Neoclassical economic theory suggests that the factors of production—land (natural resources/ natural capital), manufactured capital and labor—can be substituted for one another to a substantial degree (Hubacek and van den Bergh 2006) with extensions to include human capital (Arrow et al. 2004, Dietz et al. 2008, Engelbrecht 2009, Dasgupta in press). If these substitutions are possible then communities can be sustained if their total capital is constant, regardless of the distribution of capital across forms.

But a full sociological analysis must emphasize that the substitution of one form of capital for another is not a mechanical or automatic process but an active one (Bourdieu 1986) that often involves use of social capital, defined as the resources people access through social relations/ties (Portes 1998, Lin 2001). A sociology of sustainability, in tracing the processes of capital substitution, examines the tensions between agency and structure, as individuals, households and more aggregate actors

[3] Unfortunately, there are multiple terms for what we are calling manufactured capital, including physical capital and financial capital. For the micro-level analysis we conduct, distinctions among these terms and the attendant conceptualizations need not be explored but this is clearly an area in need of theoretical development.

develop strategies to use capital and face constraints in realizing those strategies. [4] Attention to micro-level substitutions complements the macro-level approach that dominates the current sustainability literature. [5]

4.1.1 Sustainability in a transforming economy

We focus on substitution of capital in Wolong Nature Reserve in the rapidly changing economy of China. The reserve is a source of natural capital for its human inhabitants because they make extensive use of fuelwood for heating and cooking. But that practice has substantial adverse effects on the local ecosystem, and especially the habitat of giant pandas (An et al. 2005). Electricity is available locally and can displace fuelwood use, but there are few opportunities in this very rural area to obtain the income necessary to use electricity, so a direct substitution of income for natural capital is not feasible for most households. However, the households in the Reserve also possess human and social capital and can use those resources as a basis for labor migration and wage income. This leads to the possibility of what we term "chain substitution." Local residents may be able to deploy social capital to obtain jobs that allow a return to income from their human capital, and use the income to modify their use of natural capital.

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[4] One way of enhancing well-being over the longer term is by deploying capital resources to enhance one's power and to use power to change the substitutability of one form of capital for another and thus their relative value. Because power differentials are not prominent in the context we examine here, we will not elaborate these linkages but clearly the relationships among the four capitals and the role of agency and structure in shaping access to and use of them remain undertheorized and should be a key element of a sociology of sustainability. For example, Braverman's (1974) analysis of the deskilling of labor can be viewed as a strategic effort by those with control of manufactured capital to reduce the value of human capital and thus the cost of replacing human capital with manufactured capital.

[5] Sociology has much to contribute to the existing macro-level literature as well but those considerations are beyond the scope of this Chapter.

A reframing of conventional sustainability theory is necessary to examine the processes of capital substitution in this context. At the micro-level in a capitalist economy, most individuals are not "producers" in the sense that they use manufactured capital directly to enhance their well-being.[6] Rather the dominant mode of economic activity for most households and individuals is to exchange labor based on their human capital (i.e. formal education, skills, health, physical strength) for income and use the income to purchase goods and services to support well-being. Thus at the micro-level income and wealth become the operational equivalent of manufactured capital at the macro-level. But while access to markets in which to sell labor can be taken as given for most individuals in an advanced capitalist economy, access to labor markets that allow human capital to generate income can be problematic in economies in transition.[7]

A further elaboration of our framing is the idea of chain substitution of capitals and in particular the use of social capital to gain employment and thus income (Granovetter 1973, 1985). As in many other developing regions, the local labor market in Wolong provides few opportunities for converting labor to income so

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[6] At least from Becker (Becker 1976) the concept of a household production function has been deployed to explain key aspects of human behavior. It is true that households use some manufactured capital in producing well-being (e.g. utensils in cooking, vehicles or animals in transportation). But a fundamental aspect of the shift to a capitalist economy is a shift from the household as a nexus of production *and* consumption to, for most households, the exchange of labor for income which in turn is used to procure the requisites of well-being. That is the shift towards a capitalist economy entails a shift towards a primacy of consumption over production for most members off the society. In recent history this process has continued with the commodification of many goods and services that were formally produced in the household, e.g. food preparation, child care, cleaning.

[7] In most labor markets, there are income rewards to human capital in the sense that individuals with more human capital tend to receive more income for their labor. However, as Wright (1979) demonstrated, the relationship between human capital and income depends upon social class. Again, a sociology of sustainability could profitably explore the role of class in the use of manufactured, natural, human and social capitals in the generation of well-being at both the micro and macro levels.

migration to better developed labor markets becomes critical in the chain of capital substitutions. The implications of rural to urban migration for the rural environment have been examined in a number of studies over the last two decades and most emphasize the tradeoff between using human capital locally to convert natural capital to income and moving to an urban area where labor can generate income. As young people migrate to urban areas, the supply of labor for forest clearing and the local demand for forest clearing to support agriculture both drop off (Allen and Barnes 1985, Rudel 1989, Rudel and Roper 1997, Ehrhardt-Martinez 1998, Tole 1998).

Our analysis elaborates how social capital can be used to facilitate generating income from human capital via migration to urban labor markets. We engage with the larger literature on job seeking and migration to examine the importance of social networks and in particular of "weak ties" (relationships characterized by low intimacy or infrequent interaction) in shaping migration. Thus we posit that a sociological approach to sustainability will emphasize how micro-level action to deploy capital is embedded in a social context (Granovetter 1985). As a result, social contexts can generate profound differences in how humans use natural resources (Frank et al. 2000). Indeed, one of the compelling counterargument to Hardin's (Hardin 1968) "Tragedy of the Commons" is that communities can organize in ways that prevent the overexploitation of common pool resources (Ostrom et al. 2002, Dietz et al. 2003, Dietz and Henry 2008).

4.1.2 Environmental change in contemporary China

Several recent analyses document the impact of human behavior on the environment in China (Economy 2004, Liu and Diamond 2005, Liu and Diamond

2008). While problems of industrialization, such as air and water pollution, are the most visible, local communities are placing serious strains on several critical habitats. For instance, the use of wood for cooking and heating can have substantial impact on local environment, such as our study site, the Wolong Nature Reserve (Liu et al. 2003b). More than 95% of the inhabitants are farmers living in isolated farmsteads. Their traditional livelihood depends heavily on natural capital and includes farming, fuelwood collection and livestock breeding. Fuelwood collection has been demonstrated to have an especially pronounced impact on panda habitat because the amount of fuelwood collected by inhabitants is very substantial, with a mean of over 6000 kilograms per household per year, resulting in the removal of forest canopy that provides shelter and cover for pandas (Liu et al. 1999, An et al. 2002, An et al. 2005). As a result, the panda habitat has suffered from serious degradation (Liu et al. 2001).

The inhabitants of the reserve have an alternative to obtaining energy from wood. Due to China's recent investment in hydroelectric power, electricity has become more available and more reliable. In fact, all households in the reserve have access to electricity. In our interviews, the residents of the reserve indicated they preferred electricity to fuelwood because it is more convenient, cleaner, and required less labor for gathering.[8] In contrast to electricity, fuelwood is free except for the labor required for extraction.[9] However, the cost of electricity has increased recently, in large part to offset the large government investments in producing electricity. Thus

[8] Fuelwood is not sold at the local market, and farmers in the reserve collect fuelwood mainly in winter for their own use in the following year.

[9] Although the Wolong reserve administration has developed several policies to reduce fuelwood collection, monitoring and enforcement of these policies are a problem because the settlements and fuelwood collection are very geographically dispersed.

the primary obstacle to the use of electricity is economic, and a key factor for improving economic status is through taking off-farm employment. So local residents have potential to substitute income for the use of natural capital, and would prefer to do so, but their ability to use human capital as labor to generate income locally is limited.

The move towards a market economy and urbanization forces that have affected the rest of China since the 1990's are just beginning to affect the Wolong Nature Reserve. However, some inhabitants now complement their traditional economic activities based on the use of natural capital by working in urban settings through temporary rural-urban labor migration. Previous research on labor migration has suggested that the remittance from migrants may substantially improve the livelihood of their rural households (Koc and Onan 2004, Airola 2007).

In the Wolong Nature Reserve, labor migration may have substantial impacts on the local ecosystem in several ways. First, remittances from labor migrants may be used to shift rural energy consumption from fuelwood to electricity. Second, labor migrants may also contribute to rural household economy through sending materials (e.g., food, clothes, and electronic appliances) back home, which may free up income for purchasing electricity. Third, the reduction of the local human population due to labor migration may reduce energy needs (both fuelwood and electricity) of rural households (An et al. 2001). Fourth, labor migration may reduce the labor supply for collecting fuelwood (Allen and Barnes 1985, Rudel 1989, Rudel and Roper 1997, Ehrhardt-Martinez 1998, Tole 1998). Thus the relationships among labor migration, fuelwood consumption and electricity consumption may be complex.

4.1.3 Labor migration patterns in China

For the most part, the labor migrants of the Wolong Nature Reserve, do not migrate to urban areas to settle permanently. Instead they seek temporary employment in urban settings and return to their home villages whenever needed (e.g., in planting or harvesting seasons). In this sense, they take advantage of the rapid economic development in China in seeking temporary jobs, but are not permanent urban residents in the larger urbanization process.[10] Such temporary migration is very common in China as well as many other parts of the world (United Nations 2004, Korinek et al. 2005). In the case of the Wolong Nature Reserve, in 2004, 162 people worked in cities through temporary labor migration. Although the proportion of labor migrants is small (accounting for about 6.0% of eligible laborers in the reserve), it is substantial compared with many other rural areas in China (Li and Zahniser 2002), and is increasing rapidly (Liu Mingchong, 2005, personal communication).

The determinants of labor migration and the relationship of such labor migration to macro political and economic changes have been carefully studied in contemporary China (Goldstein et al. 1997, Yang and Guo 1999, Yang 2000, Liang 2001, Li and Zahniser 2002, Fan 2003). The standard model of labor migration examines how households use human capital (e.g., gender, age and education) to generate income via labor migration (Becker 1985, Shelton and John 1996, Angrist and Evans 1998, De Jong 2000). Yet few studies have explored the impact of social capital in the context of Chinese internal migration (Zhang and Li 2003).

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[10] In some areas, income flowing from relatively permanent urban migrants back to rural villages may have important environmental consequences. This is an intriguing issue worthy of further explanation but is beyond the scope of our analysis.

It is well known that social capital may affect migration decisions (Massey 1990, Hugo 1998, Palloni et al. 2001) and facilitate migration processes (e.g., help migrants settle down and become familiar with places of destination) so that costs and risks of migration may be alleviated (Korinek et al. 2005). Social capital is also important for accessing employment information and influence (i.e. influential persons in particular labor subsectors) (Lin et al. 1981, Granovetter 1995, Bian 1997, Yakubovich 2005). It is important to differentiate the strength of social ties as ties with different strengths may have different roles in facilitating labor migration. For instance, relatives may be perceived as stronger ties than friends, while friends may be perceived as stronger ties than acquaintances (Granovetter 1995, Bian 1997). Strong social ties may be more reliable in facilitating migration processes such as transportation and settlement, while weak social ties may expand information about employment opportunities (Massey and Espana 1987, Massey and Espinosa 1997, Wilson 1998). Moreover, weak ties may provide direct access to influence, while strong ties are usually indirectly associated with influence (Granovetter 1995, Bian 1997, Yakubovich 2005).

Granovetter (1973; 1995) explained how ties, in particular weak ties, might affect employment seeking. His arguments are very salient for the reserve. The Wolong Nature Reserve is in a mountainous rural area, and is far from any urban areas (>100 km), which makes communication between the reserve and the outside difficult. Without any government institutions or other formal organizations providing employment information, social capital is an important source of such information. Without social capital it may be very difficult for a household to use human capital to generate income.

To understand the environmental impact of labor migration in the reserve, we must retrace a causal path that starts with the use of fuelwood, from there back to the economic and demographic impacts of labor migration, and finally from labor migration to the social capital that facilitates such migration. Our model is summarized in Figure 4.1. Building on the classic effects of social ties on labor outcomes, our first hypothesis is that access to social capital, especially weak social ties, facilitates labor migration. The second hypothesis is that labor migration reduces household fuelwood consumption, as income is substituted for the use of natural capital. Thus the resources individuals access through social relations indirectly affect fuelwood consumption. Our analysis elucidates the links between social relations, labor migration and fuelwood consumption and thus shows the processes by which one form of capital is substituted for another.

4.2 METHODS

4.2.1 Household surveys

Our in-person interviews were conducted from May to August 2005 in the Wolong Nature Reserve. We chose household heads or the spouses of household heads as interviewees because they are usually the decision makers on household affairs and know the most about other household members' information (e.g., employment and income). From the government's household registration list containing about 1200 households in all the groups in the reserve (groups are nested within villages within townships in rural China), households with temporary labor migrants were identified by group heads (farmers who are elected by their group members to coordinate some group affairs such as recruiting laborers for group infrastructure work). There were 138 households with temporary labor migrants in 2004. No eligible respondent in 7 of these households could be reached within 5 revisits and data from 2 households were not complete, which resulted in 129 households corresponding to 152 labor migrants. For the purpose of comparison, we also interviewed 215 households out of 223 households randomly selected from 1018 households that were not identified by group heads as households with labor migrants. Our overall response rate for interviews was 95%.

We collected socio-demographic information on individual members and economic, social ties and fuelwood consumption data for households. We asked the average amount (weight) of daily fuelwood consumption in the previous year for both the winter season when more fuelwood is needed and the summer season when less fuelwood is needed. Household fuelwood consumption was therefore measured as a summation of daily consumption across the year. As noted above, previous studies in this reserve have identified fuelwood collection as one of the main reasons of the degradation in the local ecosystem (Liu et al. 1999, Liu et al. 2001, An et al. 2002, An et al. 2005).

In households without any labor migrants, we asked respondents about their social ties with people who were living or working (including temporary migrants) in cities outside the reserve. Since labor migration could lead to social ties, in households with labor migrants we asked respondents to recall their social ties before migration. We measured the strengths of social ties with relatives taken as strong ties, acquaintances as weak ties, and friends as ties of moderate strength. We also asked if each type of their ties includes people holding leadership positions. Our measures of social ties in households with labor migrants are retrospective of the pre-migration social network. Although accurate recall of social ties is difficult (Bernard et al. 1984, Marsden 1990), people tend to report social ties with whom they have more interactions (Neyer 1997, Feld and Carter 2002) and hence are more important for activities such as labor migration. We used dummy variables to denote the availability of various social ties because measures of network size tend to be biased in retrospective studies (Brewer 2000). Wolong Nature Reserve is a relatively isolated area where inhabitants do not have many ties to the outside, so dichotomous measures of social ties still capture most of the variation in social resources among households. In households with labor migrants, we asked how much remittance labor migrants send back home.

4.2.2 Causal inference

Because of self-selection into labor migration, the relationship between migration and fuelwood consumption may be confounded with other factors. In the absence of a randomized or natural experiment assigning people to migrate or not, any estimated effect of labor migration on an outcome may be spurious. This is reflected in the fundamental counterfactual question: "How much fuelwood would a household with labor migrants have consumed if the household member(s) had not temporarily worked outside of the reserve?" This question is counterfactual because we cannot observe the fuelwood consumption of households with labor migrants under the condition of no one working outside of the reserve. Neglect of this self-selection process can result in invalid inferences (Winship and Morgan 1999, Hirano and Imbens 2002).

We approximate counterfactual conditions using propensity score weighting (Rosenbaum and Rubin 1983, Robins and Rotnitzky 1995, Hirano and Imbens 2002, Hirano et al. 2003, Morgan and Harding 2006). Propensity score techniques use the logic of comparing individuals in the treatment group (in our case, the treatment group is composed of the households with temporary labor migrants) to individuals in the control group (households without labor migrants) with a similar propensity score (likelihood of working outside). The propensity score is defined as (Rosenbaum and Rubin 1983):

$$e(x) = \Pr(m = 1 \mid x), \tag{4.1}$$

where m is a dummy variable indicating treatment (i.e. 1 if one or more members of a household were working outside the reserve; 0 otherwise); $e(x)$ is the propensity for receiving the treatment and can be estimated using a logistic regression model using covariates x (e.g., household level human capital and economic conditions, geographical information, and social capital).

We use weights based on the propensity scores in estimating the average causal effect of labor migration on fuelwood consumption (Robins and Rotnitzky 1995, Hirano and Imbens 2002, Hirano et al. 2003). The weights are defined by

$$\omega(m, x) = \frac{m}{e(x)} + \frac{1-m}{1-e(x)}. \tag{4.2}$$

Therefore, a household with migrants is weighted by $1/e(x)$ and a household without migrants is weighted by $1/(1-e(x))$. In other words, the lower the propensity of having migrants for those households with labor migrants, the greater weight they are given. Similarly, the higher the propensity of having migrants for those households without migrants, the more weight they are given. In this way, the estimation of the average causal effect focuses mainly on the strongest overlap in propensity, those with lower propensity in the treatment group and those with higher propensity in the control group (Figure 4.2).

The weighting in (Eq. 4.2) is informative for policy considerations because it reflects individual responses to incentives. If policies focus on changing incentives and resources for labor migration, then estimates of effects should focus on those most likely to respond to changes in policies: those who were employed outside the reserve but who had low propensity for doing so and, therefore, might not have become employed outside the reserve if there were fewer incentives for doing so; and those who were not employed outside the reserve but who had high propensity for doing so and therefore might respond to increases in incentives. Thus, the estimate using the weights in (Eq. 4.2) is referred to as the effect of the treatment for people at the margin of indifference (EOTM) (Heckman 2005).

Propensity scores can also be used as a basis for matching or defining strata (Rosenbaum and Rubin 1983, Morgan and Harding 2006). We prefer the weighting approach because (1) the weighting scheme is relatively simple and intuitive; (2) estimates using the weights are easy to obtain (e.g., using weighted least squares) and can be implemented within the context of simple or more complex models; (3) because the estimand is a smooth function of the data (as in the weighted regression), bootstrapping techniques can be employed to calculate standard errors that reflect uncertainty in estimating the propensity; and (4) all subjects contribute to the analysis (though not equally, by definition). In fact, all matching estimators can be considered examples of weighting approaches (Morgan and Harding 2006), but only the Robins' approach we use here has been proven to improve the efficiency of estimation (Hirano et al. 2003). The greatest concern about weighting is that extreme weights could exert undue influence on the estimates. This is easily addressed by examining the distribution of weights and trimming extreme values.

Under some circumstances, separate causal effects for the migration group and the non-migration group are of interest. These estimates can be obtained with minor changes to the weights. In particular, to estimate the effect of labor migration for those households in which a member was working outside of the reserve, the following weights can be used:

$$\omega_{mi}(m,x) = m + (1-m)\frac{e(x)}{1-e(x)}. \qquad (4.3)$$

Thus those working outside of the reserve are weighted with a value of one, and members of the comparison group are given more weight if they have a higher propensity to migrate. As a complement, to estimate the effect of temporary labor migration for those households in which no one was working outside the reserve, the following weights can be used:

$$\omega_{nonmi}(m,x) = m\frac{1-e(x)}{e(x)} + (1-m). \qquad (4.4)$$

Here those households in which no one was working outside the reserve are assigned a weight of one, and those with labor migrants are given more weight if they have a lower propensity to migrate.

4.2.3 Analytical approach

We first model the propensity for labor migration as a function *inter alia*, of social ties. Then we estimate the effect of labor migration on fuelwood consumption. All laborers (912 people) from the 344 households that we interviewed are used in logistic regression models to estimate the propensity for labor migration. Based on past studies of labor migration in China (Goldstein et al. 1997, Yang and Guo 1999, Yang 2000, Liang 2001, Li and Zahniser 2002, Fan 2003, Zhang and Li 2003), we chose both individual level and household level factors as potential determinants of temporary labor migration. At the individual level, we chose gender, age, marital status, education level, number of children younger than 15 years of age and availability of extended household member. At the household level, we chose amount of land, non-migration income (measured by excluding migration income from total household income), number of laborers (18~60 years of age, people beyond this range usually do not work outside) and indicator of township the household is located. We extend the model specifications suggested in the literature by adding social capital to these individual and household level human capital and income and wealth factors. We did not find multicollinearity among independent variables.

The first model includes three dummy variables denoting the availability of relatives, friends and acquaintances living or working in cities outside the reserve. In models two through four, we isolate effects of each particular tie as well as the extent to which the ties hold leadership positions. The fifth model controls for the availability of any type of social ties (aggregation of the availability of three types of social ties) and any ties to people holding leadership positions (aggregation of the availability of three types of social ties holding leadership positions) outside the reserve. We use the last model to calculate the propensity weights because it has the best fit according to AIC criterion (the model having the lowest AIC is the best). Moreover, because this model includes the primary factors predicting labor migration described in the literature as well as measures of social capital we use it as a basis of causal inference in the absence of a randomized experiment which would have been ethically and logistically difficult to conduct in this context (Shadish et al. 2002). After calculating the propensity weights, we confirm that the weights achieve balance on our covariates by testing for differences between households with and without temporary labor migrants using the weighted and unweighted data. Reduction in differences when employing the weights suggests that selection bias has been adjusted via the weights (Morgan and Harding 2006).

Next, we use the estimated propensities to weight a standard regression of the effect of labor migration on fuelwood consumption. As noted above, the weights take into account factors affecting the propensity of labor migration and allow us to focus on the estimated effect of those on the margin of deciding whether work in an urban context (Heckman 2005). In addition to the migration status of households (measured with a indicator of whether a member of the household had engaged in temporary labor migration in 2004), we control for household size, availability of senior members (people over 60 years of age), household income, amount of land, number of pigs the household breed, and indicator of township the household is located as covariates in fuelwood consumption models as suggested by past studies in this reserve (An et al. 2001, An et al. 2002). Moreover, a few determinants of labor migration that are not well balanced through the propensity weighting are also controlled as covariates. We hypothesize that there are effects of labor migration on fuelwood consumption beyond the direct economic returns from labor migration because migration reduces household labor for gathering fuelwood and decreases demand as a result of the absence of a household member. In addition, migrants may also send materials (e.g., food, clothes, and electronic appliances) home. To reflect the potential indirect effects of labor migration, total household income, as an alternative to non-migration income, is accounted for in some fuelwood consumption models.

We use all working age individuals (household members from 18 to 60 years of age) as units of analysis in estimating the propensity model because it is individuals who choose whether work outside the reserve. Since we also use some household predictors (e.g., amount of land owned by the household) in this model, we report robust standard errors in the propensity models to account for the resulting lack of independence among observations.[11] We analyze fuelwood consumption at the *household* level because fuelwood is consumed by households. The highest propensity score of any individual in the household is assigned to the household in that few households had more than one labor migrant.

Because there is uncertainty in the estimates of the propensity scores that are used in weighting our fuelwood consumption models, we use case based bootstrapping to calculate standard errors (Efron and Tibshirani 1993). For each

[11] We corrected for dependencies among members of the same household with Huber's variance correction in STATA 8.0 (STATA Corp., College Station, TX, USA).

estimate of the propensity and fuelwood consumption models, we calculate standard errors from 500 bootstrap replicates that are then the basis for t-ratios.

4.2.4 Quantifying the robustness of the inference

Although we have attempted to reduce bias in our estimate by controlling (through propensity score weighting) for many well recognized factors affecting labor migration and fuelwood consumption as well as drawing on our own understanding of the phenomenon in the reserve, we may have omitted confounding factors that could bias our estimates. Therefore, we explore the robustness of our inferences to the possibility of omitted variables.

Our approach to quantifying robustness can be considered an extension of sensitivity analysis (Rosenbaum and Rubin 1983, Holland 1989, Copas and Li 1997, Robins et al. 2000, Scharfstein and Irizarry 2003). Sensitivity analyses consider a set of possible estimates given a broad set of alternative conditions. As in sensitivity analysis, we consider how violations of assumptions could affect estimates. But rather than reporting how violations of assumptions produce a range of estimates, we focus on exactly how much an assumption must be violated to invalidate an inference. As a result, the indices quantify the robustness of the original inference.

Classically, internal validity can be expressed in terms of confounding variables that are correlated with both the predictor of interest and the outcome (Shadish et al. 2002). We express the robustness of our inferences to these two relationships by employing the impact threshold (Frank 2000, Pan and Frank 2003). Frank (2000) defines the impact of a confounding variable on an estimated regression coefficient as $r_{vy} * r_{vm}$, where r_{vy} is the correlation between a covariate, v, and the outcome y; and r_{vm} is the correlation between v and m, the predictor of interest (for example, m is an indicator of the status of labor migration of the household -- see Figure 4.3).

To obtain the impact necessary to invalidate an inference, define $r^{\#}$ as a quantitative threshold for making inferences. Note that there is a direct relationship between r (the observed correlation between the predictor of interest and the outcome) and the statistical significance (t-ratio) of the predictor of interest (Cohen and Cohen 1983), $t = \frac{r\sqrt{d.f.}}{1-r^2}$, where $d.f.$ is the degree of freedom in the regression analyses. In particular, defining $t_{critical}$ as the critical value of a t-distribution (e.g., for p $\leq$0.05), then $r^{\#} = \frac{t_{critical}}{\sqrt{d.f.+t^2}}$ defines a threshold based on statistical significance. That is, r_{my} (the correlation between the predictor of interest, m, and the outcome y) will be statistically significant if and only if it is greater than $r^{\#}$.

Given the definition of $r^{\#}$, a simplification of Frank (2000) shows that the original inference from a bivariate regression is invalid if $r_{vy} * r_{vm} > \frac{r_{my} - r^{\#}}{1-|r^{\#}|}$. Thus the quantity $\frac{r_{my} - r^{\#}}{1-|r^{\#}|}$ defines the impact threshold for a confounding variable for simple linear regression. That is, if the impact ($r_{vy} * r_{vm}$) of a confounding variable is greater than $\frac{r_{my} - r^{\#}}{1-|r^{\#}|}$ the original inference is not valid. The corresponding threshold for the multivariate case with covariates z is $\frac{r_{my \cdot z} - r^{\#}}{1 - |r^{\#}|} \sqrt{(1 - r_{mz})^2 (1 - r_{yz})^2}$.

Critically, because the impact is defined by correlation coefficients it can be readily understood by social scientists comfortable with correlation and the general linear model. This makes it an ideal complement to our use of propensity score weighting applied in a general linear model.

4.3 RESULTS

4.3.1 Data summary

There were 152 labor migrants from 129 households, and they worked in the construction (31.6%), transportation (11.8%), industry (18.4%), service (29.6%) and business (7.9%) sectors. About 74.3% of them worked in cities within the Sichuan province, and 25.7% of them worked in cities in other provinces in China. Summary statistics of individual level variables are presented in Table 4.1. About 52% of 912 laborers in our sample of 344 households were male with an average age of 36 years, and 79.1% of the laborers were married. The mean number of years of education was 6. On average, each of these laborers had less than one child under 15 years, and about half the laborers lived with extended household members such as parents or parents-in-law. Labor migrants accounted for 16.7% of 912 laborers in our stratified sample of 344 households.

At the household level, the mean household size was 4.7 people, while the mean number of laborers was 2.7 (Table 4.2). About 1/3 of these households had senior members. The mean non-migration income was 10.253 thousand Yuan, and the mean total household income was 11.377 thousand Yuan. On average, each household owned 0.282 hectares of cropland, breed about 3 pigs, and about 60% of these households were located in the Gengda township. About half of the households had relatives working or living in urban areas, but less than half of these relatives held leadership positions (Table 4.2). Only about 19% and 24% of the households had friends and acquaintances working or living in urban areas respectively, and few of these ties held leadership positions. More strong ties (i.e. relatives) were reported than weak ties (i.e. acquaintances) presumably because people tend to report social ties with whom they have more interactions (Neyer 1997, Feld and Carter 2002). By combining different types of social ties, 66.3% of the households had social ties in urban areas and 30.2% of the households had ties with people holding leadership positions. The proportion of households with labor migration in the overall study area was 11.9% but in our stratified sample 37.5% of households had labor migrants. On average, each household in our sample consume 6325 kilograms of fuelwood.

4.3.2 Determinants of labor migration

Models of the determinants of temporary labor migration are presented in Table 4.3. Model 1 shows that households with weak ties (i.e. acquaintances) were significantly ($p \leq 0.001$) more likely to have labor migrants than were other households—this form of social capital facilitates being able to use human capital to produce income. The effect of relatives, representing strong ties, is not statistically significant. These results are consistent with Granovetter's "the strength of weak ties" hypothesis (Granovetter 1973, 1995). Holding all other factors constant, the availability of an acquaintance increases the odds of labor migration by 2.54, while the effects of the availability of relatives and friends on labor migration do not significantly differ from zero (see model 1 in Table 4.3). When exploring different types of social ties separately controlling for demographic and economic factors as covariates (see models 2 through 4), the availability of relatives and friends working or living in urban areas does not have significant effects on labor migration, while the availability of acquaintances still has a significant ($p \leq 0.01$) positive effect on labor migration with a similar magnitude as that in model 1. Moreover, the insignificance of ties holding leadership positions indicates that leadership ties were not more helpful than non-leadership ties. When different types of ties were combined (model 5; AIC = 587.629, pseudo $R^2 = 0.319$), the availability of social ties in cities outside of the reserve significantly ($p \leq 0.01$) increased people's probability of labor migration. Holding all other factors constant, the availability of social ties increased the odds of labor migration by 2.21. Thus social capital is, as expected, very important in obtaining income from labor.

Human capital and economic conditions had similar effects across the 5 models. It is not surprising that human capital was very important. Men were more likely ($p \leq 0.001$) to work outside the community than women because men are usually expected to assume economic responsibilities for households in rural areas of China. The odds of labor migration for men is about 2.82 times higher than that for women (see model 5). Both age and its quadratic term had significant ($p \leq 0.05$) effects on labor migration. The quadratic relationship between age and migration shows that the probability of migration increases until 30 years and then declines as age increases. The odds of labor migration for married people is only about 0.21 times of that for unmarried people. Education increases the probability of labor migration significantly ($p \leq 0.001$). Holding all other factors constant, the odds of labor migration increased by a factor of 1.20 for each additional year of education. No effects of extended household member(s) and the number of children under 15 years were detected. The number of laborers in the household had a significant ($p \leq 0.05$) positive effect on labor migration. Each additional household laborer increased the odds of labor migration by 1.39 holding other factors constant (Model 5 in Table 4.3). Amount of cropland of the household did not have a significant effect on labor migration. Non-migration income had a significant ($p \leq 0.01$) negative effect on labor migration. Holding all other factors constant, the odds of labor migration decreased by a factor of 0.93 with an increase in non-migration income of one thousand Yuan. These effects are consistent with the fact that labor migration is a way of finding alternative opportunities for those with the most limited opportunities in the reserve to balance the inequalities in economic status and demographic conditions among households. Finally, residing in Gengda township had a significant ($p \leq 0.01$) positive effect on labor migration. The odds of labor migration for people in the Gengda township was 2.29 times of that for people in the Wolong township. This result reflects the fact that the Gengda township is geographically closer to urban areas outside the reserve so its inhabitants have access to more information and material exchanges with the outside than those living in the Wolong township. Our results of the determinants of temporary labor migration are consistent with many other empirical studies at regional or national levels in China (Goldstein et al. 1997, Yang 2000, Li and Zahniser 2002, Fan 2003).

4.3.3 Balancing covariates using the propensity score weights

We tested for differences between those households with labor migrants and those without labor migrants on household level variables that are used in the propensity model. Note that we tested only for the household level characteristics because our next model of fuelwood consumption is defined at the household level. Test statistics with and without using propensity weighting as in equation (2) are presented in Table 4.4. Propensity weighting reduced the differences between the migration group and the non-migration group on almost all the household level variables except the Gengda township indicator. Land, non-migration income, and number of laborers in the household were not significantly different between the migration and non-migration groups after weighting. Although the availability of social ties outside of the reserve and that of ties to those holding leadership positions were still higher for the migration group than those for the non-migration group in the weighted analysis, there was less difference between the two groups after weighting. These two social ties covariates and the Gengda township indicator were still significantly different between migration and non-migration groups after weighting, and therefore were controlled in estimating the effects of labor migration on fuelwood consumption.

4.3.4 Estimation of the effects of labor migration on fuelwood consumption

Estimates of the effect of labor migration on fuelwood consumption with propensity weighting are presented in Table 4.5. Fuelwood consumption of households with labor migrants was significantly less than fuelwood consumption of households without migrants. When non-migration income, together with other covariates, was included in the model, households with migrants consume 1827 kilograms less fuelwood (~28.9% of average annual household fuelwood consumption in the reserve) on average than those without migrants ($p \leq 0.001$). In contrast, the effect of labor migration without using weights was estimated to be 1647 kilograms ($p \leq 0.001$).

We also estimated the effects of labor migration separately for the migration groups and non-migration groups using estimate-specific weights (see equations 3 and 4). Labor migration had less effect on reducing fuelwood consumption for those households in the migration group (the 3rd row in the 1st column of Table 4.5), while the effect is strongest for those in the non-migration group (the 4th row in the 1st column of Table 4.5). Presumably, the difference is due to the differences in characteristics between these two groups. For example, a high propensity of labor migration may indicate that the household has more laborers, and a reduction of one laborer from a household that has many laborers may not affect as much the supply of labor for fuelwood collection as that from a household that has few laborers.

In addition to the direct economic contribution of labor migration, following the deforestation literature we also hypothesized indirect effects of labor migration on fuelwood consumption. To estimate these effects non-migration income was replaced with total household income with results reported in the 2nd column of Table 4.5. In this model, the effect of labor migration is net of the income it contributes to the household. Labor migration still has significant negative effects on fuelwood consumption, although the magnitude of effects is smaller than that when non-migration income was controlled for (1st column of Table 4.5). This result suggests that labor migration has both a direct economic contribution and an indirect effect on reducing fuelwood consumption. The indirect effect may occur because migrant laborers send materials (e.g., food, clothes, and electronic appliances) home, and their absence may reduce both the need for fuel in the household and the labor available to gather fuelwood, and may even affect the lifestyles of their household (e.g., using electric stoves and other appliances which in turn may make electrical use routine for heating as well).

4.3.5 The robustness of the inference

We base the analysis of the robustness of the inference on the estimate of the average effect of labor migration, including non-migration income (coefficient = -1827, standard error = 242). The observed t-ratio of -7.55 translates to a correlation coefficient of -0.378 and, for a sample size of 344, the threshold for statistical significance ($r^{\#}$) is a correlation of -0.107. The corresponding impact threshold is -0.25. That is, to invalidate the inference the magnitude of the impact of an unmeasured confounding variable must be greater than 0.25. Furthermore, the magnitude of r_{vy} (the correlation between the unobserved confounding variable and fuelwood consumption) must be greater than 0.47 and the magnitude of r_{vm} (the correlation between the unobserved confounding variable and labor migration) must be greater than 0.54 to invalidate the inference.[12] Each component correlation is large by social science standards (Cohen and Cohen 1983). Moreover, these are zero-order correlations, assuming that the unmeasured confounder is uncorrelated with the

[12] The zero order correlations are not necessarily equal when the impact is maximized with covariates in the model. If the component correlations do not take these exact values then the impact would have to be greater than .22 to invalidate the inference.

measured covariates (Frank 2000). The relevant partial correlations (Cohen and Cohen 1983) from which the impact of an unobserved confounder would be constructed would be smaller than the zero-order correlations because of correlations with existing covariates.[13]

Though the magnitude of the impact threshold for an unmeasured variable can be interpreted in terms of typical patterns of correlation in the social sciences, it is also helpful to compare the threshold to the impacts of measured covariates. Based on zero order correlations, the magnitude of the impact of the indicator of township (Gengda versus Wolong) is the largest of the existing covariates. Its impact is -0.037 and the sign is in the direction that reduces the negative effect of labor migration on fuelwood consumption. Thus the magnitude of the impact of an unmeasured confound necessary to invalidate the inference that labor migration affects the amount of fuelwood consumed in the household (0.25) would have to be more than six times greater than the magnitude of the strongest impact of the measured covariates, -0.037.

4.4 CONCLUSIONS

We have suggested that an appropriate sociological approach to sustainability is to consider the strategies individuals and households deploy to generate well-being from their income and wealth, access to natural capital, human capital in labor and social capital. This approach is consistent with the existing sustainability literature that emphasizes problems of capital substitution. But it extends that approach to include the important sociological insight of the tension between agency, in the form

[13] Frank (2000) refers to this as absorption of the impact of an unmeasured confound by existing covariates.

of individual and household strategies, and structural constraints, in the form of limited access to some forms of capital.

In the Wolong Nature Reserve, the most crucial environmental threat is deforestation and the resultant degradation of panda habitat. On average, local residents use very substantial amounts of natural capital in the form of fuelwood for cooking and winter heating. While the possibility of substituting electricity for fuelwood exists, the costs of electricity and the paucity of local opportunities to convert human capital, via labor, into income preclude this move away from the use of natural capital for most households—a structural constraint. However, our analysis shows that a form of social capital, weak ties, is often used to gain access to extra-local employment and that the income from this employment then displaces the use of local natural capital. Of course the electricity generation from hydropower plants and coal burning may also have negative environmental impacts, which are beyond the scope of this study.

In addition to providing a "demonstration of concept" for our proposed sociological approach to sustainability, our results also address two other issues in the literature. First, we have replicated in rural China a finding developed elsewhere—that among forms of social capital it is weak ties that matter most in finding opportunities to find employment (Garip 2008, Pfeffer and Parra 2009). We note that strong ties may produce weak ties, but in our research and that of others back to Granovetter (1973), it is weak ties that have the most impact.

Second, we have shown that, at least in the context of the Wolong Nature Reserve, the effect of labor migration on deforestation comes from the ability to use increased household income to purchase a substitute for local natural capital. It is well understood that labor migration can have substantial environmental consequences (Bilsborrow and Ogendo 1992, Bilsborrow 2002, Rudel et al. 2002, Aide and Grau 2004, Liu and Diamond 2005), an issue first raised by Marx (Foster 1999). But without in-depth understanding of how migration decisions are shaped by context and why they vary across individuals and households, it is hard to understand the dynamics and impacts of migration and ultimately the environmental consequences of migration (Walker 2008). In Wolong context, our results contrast with some earlier findings on labor migration and deforestation that emphasize the loss of labor supply as the mechanism by which extra-local employment eases deforestation (Allen and Barnes 1985, Rudel 1989, Rudel and Roper 1997, Ehrhardt-Martinez 1998, Tole 1998).

The overall adverse effects of Chinese economic development are well documented (Liu and Diamond 2005, Liu and Diamond 2008) and by 2015 China is projected to have, after the U.S., the second largest ecological footprint of any nation (Dietz et al. 2007). Policy efforts to ameliorate this impact and move China and other economies in transition towards a more sustainable path must be designed with sensitivity to local context to avoid perverse effects (Liu et al. 2007a). The effects of weak ties in the Wolong Reserve communities suggest a relatively low-cost mechanism to encourage the substitution of income for use of local natural capital. In the reserve, it appears that access to extra-local labor markets is the key structural constraint on household strategies. Creating local labor markets that allow exchange of labor for income is difficult and the ability to do so without violating the sustainability goals of the Nature Reserve may be limited. However, enhancing social capital by providing better information on and access to extra-local labor markets is a relatively low cost policy option for government. In the case of Wolong this could reduce the demand on fuelwood. However, the effects of reducing this structural constraint on deploying human capital to produce income is context specific and so might or might not reduce the use of local natural capital in other contexts. Developing effective policies requires careful analysis of how those influenced by the policies will respond.

Finally, while we have emphasized the household and individual as units that deploy capital to enhance their well-being, a sociology of sustainability should not limit itself to the micro level. Part of the sociological tradition is to consider not only individuals and households as agents but also communities, social movements, formal organizations, government and nations. Sociology could contribute fruitfully to our understanding of sustainability by examining the strategies used by these collective actors and the constraints they face in deploying the capital resources available to them.

Table 4.1. Descriptive statistics of individual level variables.

Independent Variables	Mean (Standard Deviation)
Male (male = 1 and female = 0)	0.520 (0.500)
Age (years)	36.034 (11.541)
Age Squared	1431.492 (889.042)
Married (married = 1 and single = 0)	0.791 (0.407)
Education (years)	5.998 (3.490)
Children (number of children with age $\leq$ 15 years)	0.867 (0.933)
Extended (1 if there is extended member in the household; 0 if no extended member in the household)	0.507 (0.500)
Migrant (1 if the individual is a labor migrant; 0 if the individual is not a labor migrant)	0.167 (0.373)

(n = 912)

Table 4.2. Descriptive statistics of household level variables.

Variables	Mean (Standard deviation)
Household Size (number of people in the household)	4.663 (1.288)
Laborers (number of working age people—18~60 years of age—in the household)	2.651 (1.061)
Senior (1 if there is senior member in the household; 0 if no senior member in the household)	0.326 (0.469)
Non-migration Income (thousands of Yuan)	10.253 (7.887)
Total Household Income (thousands of Yuan)	11.377 (9.376)
Land (hectares)	0.282 (0.152)
Pigs (number of pigs the household breed)	2.881 (2.159)
Gengda (Gengda township = 1 and Wolong township = 0)	0.599 (0.491)
Relative (1 if there is relative outside the reserve; 0 if no such relative)	0.517 (0.500)
Relative Leader (1 if there is relative outside the reserve holding leadership position; 0 if no such relative)	0.215 (0.412)
Friend (1 if there is friend outside the reserve; 0 if no such friend)	0.189 (0.392)
Friend Leader (1 if there is friend outside the reserve holding leadership position; 0 if no such friend)	0.061 (0.240)
Acquaintance (1 if there is acquaintance outside the reserve; 0 if no such acquaintance)	0.244 (0.430)
Acquaintance Leader (1 if there is acquaintance outside the reserve holding leadership position; 0 if no such acquaintance)	0.055 (0.229)
Tie (1 if there is any type of social tie outside the reserve; 0 if no tie outside the reserve)	0.663 (0.473)
Tie Leader (1 if there is any type of social tie outside the reserve holding leadership position; 0 if no such tie)	0.302 (0.460)
Migration (1 if there is labor migrant(s) in the household; 0 if no labor migrant(s) in the household)	0.375 (0.485)
Fuelwood Consumption (kilograms)	6325 (4499)

(n = 344)

Table 4.3. Determinants of labor migration models.

Independent variables	Coefficient (Adjusted standard error) [odds ratios]				
	Model 1	Model 2	Model 3	Model 4	Model 5
Male	1.029*** (0.222) [2.798]	1.037*** (0.214) [2.821]	1.034*** (0.217) [2.812]	1.044*** (0.223) [2.841]	1.035*** (0.222) [2.815]
Age	0.316** (0.117) [1.372]	0.320** (0.120) [1.377]	0.315** (0.121) [1.370]	0.325** (0.118) [1.384]	0.300* (0.121) [1.350]
Age Squared	-0.005** (0.002) [0.995]	-0.005** (0.002) [0.995]	-0.005** (0.002) [0.995]	-0.005** (0.002) [0.995]	-0.005** (0.002) [0.995]
Married	-1.725*** (0.344) [0.178]	-1.540*** (0.340) [0.214]	-1.644*** (0.335) [0.193]	-1.726*** (0.339) [0.178]	-1.563*** (0.361) [0.210]
Education	0.186*** (0.043) [1.204]	0.181*** (0.040) [1.198]	0.182*** (0.041) [1.200]	0.190*** (0.043) [1.209]	0.186*** (0.044) [1.204]
Children	0.072 (0.182) [1.075]	0.110 (0.184) [1.116]	0.091 (0.184) [1.095]	0.066 (0.182) [1.068]	0.067 (0.187) [1.069]
Extended	0.314 (0.312) [1.369]	0.417 (0.311) [1.517]	0.316 (0.318) [1.372]	0.287 (0.315) [1.332]	0.346 (0.315) [1.413]
Laborers	0.359** (0.126) [1.432]	0.336** (0.121) [1.399]	0.325** (0.124) [1.384]	0.348** (0.126) [1.416]	0.327* (0.129) [1.387]
Land	-0.267 (0.870) [0.766]	-0.108 (0.816) [0.898]	0.016 (0.863) [1.016]	-0.300 (0.826) [0.741]	-0.160 (0.859) [0.852]
Non-migration Income	-0.082*** (0.024) [0.921]	-0.069** (0.022) [0.933]	-0.065** (0.022) [0.937]	-0.079*** (0.024) [0.924]	-0.073*** (0.023) [0.930]
Gengda	0.782*** (0.238) [2.186]	0.907*** (0.235) [2.477]	0.937*** (0.238) [2.552]	0.793*** (0.238) [2.210]	0.830*** (0.246) [2.293]
Relative	0.196 (0.217) [1.217]	-0.015 (0.263) [0.985]			
Relative Leader		0.560 (0.323) [1.751]			
Friend	0.197 (0.281) [1.218]		0.061 (0.333) [1.063]		
Friend Leader			0.710 (0.513) [2.034]		
Acquaintance	0.930*** (0.244) [2.535]			0.892*** (0.274) [2.440]	
Acquaintance Leader				0.307 (0.384) [1.359]	
Tie					0.795** (0.294) [2.214]
Tie Leader					0.316 (0.268) [1.372]
Intercept	-8.003*** (1.959)	-8.045*** (1.958)	-7.848*** (1.991)	-8.009*** (1.956)	-8.126*** (2.014)
AIC	589.247	599.763	600.832	588.033	587.629
Pseudo R^2	0.319	0.304	0.303	0.319	0.319

Significance: * $p \le 0.05$; ** $p \le 0.01$; *** $p \le 0.001$ (two-tailed tests). n = 912 ($n_{mi} = 152$)

Table 4.4. Testing for balance between migration group and non-migration group with and without propensity weighting.

	Migration group (n = 129)	Non-migration group (n = 215)				
Variable	Mean (Standard deviation)		t-ratio [a] (unweighted)	χ^2 (unweighted)	t-ratio (weighted)	χ^2 (weighted)
Land	0.286 (0.153)	0.279 (0.152)	-0.41		-0.39	
Non-migration Income	9.098 (7.796)	10.945 (7.878)	2.11		0.37	
Laborers	3.093 (1.169)	2.386 (0.894)	-5.91		-1.58	
Gengda	0.698 (0.461)	0.540 (0.500)		8.39		10.30
Tie	0.814 (0.391)	0.572 (0.496)		21.10		15.95
Tie Leader	0.426 (0.496)	0.228 (0.420)		15.05		8.29

[a] Positive value indicates households with labor migrant(s) have lower mean than those without labor migrant(s).

Table 4.5. Estimated effect of labor migration on fuelwood consumption (kilograms) using general linear models (GLM).

Models	Coefficient (Bootstrap standard error)	
	Covariates including non-migration income	Covariates including total household income
GLM: unweighted	-1647***	-1262**
	(467)	(461)
GLM: average effect of labor migration	-1827***	-1482***
	(242)	(249)
GLM: effect for migration group	-1253**	-988*
	(424)	(409)
GLM: effect for non-migration group	-2067***	-1668***
	(263)	(279)

Significance: * $p \leq 0.05$; ** $p \leq 0.01$; *** $p \leq 0.001$ (two-tailed tests). n = 344

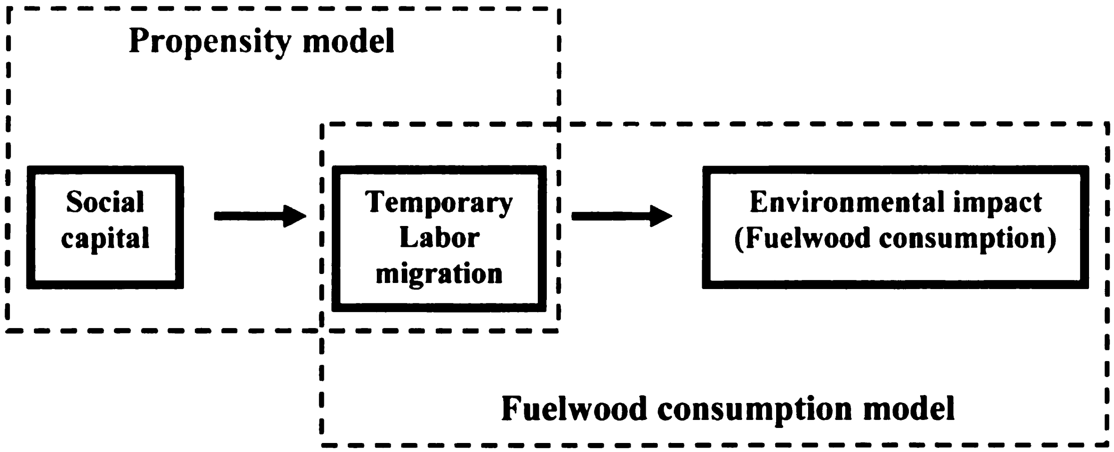

Figure 4.1. The effect of social capital on the environment as mediated by labor migration.

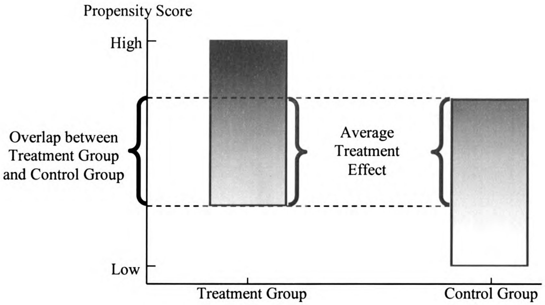

Figure 4.2. Overlap in propensity scores between treatment group and control group.

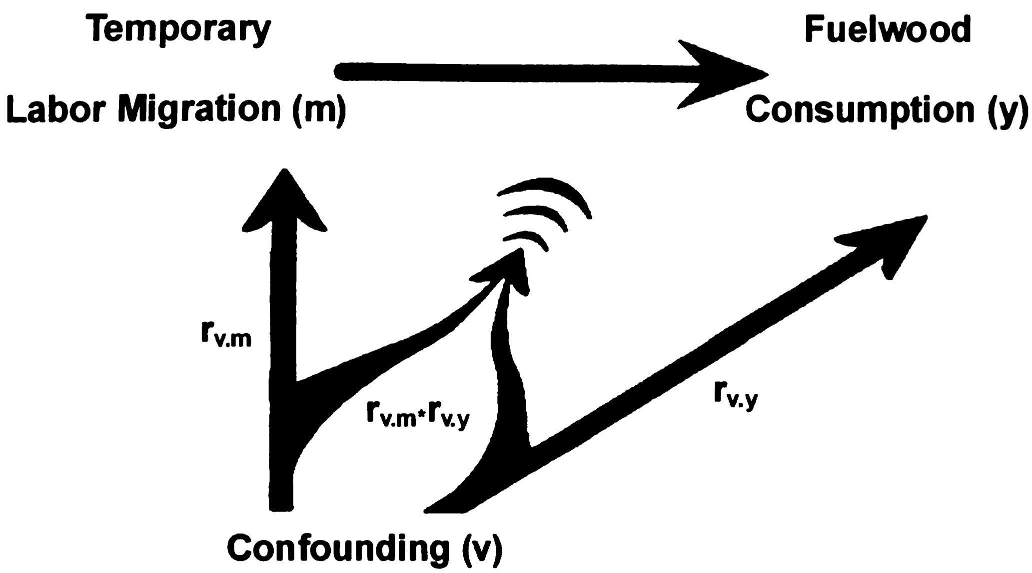

Figure 4.3. The impact of a confounding variable on a regression coefficient.

CHAPTER 5

MODELING THE EFFECTS OF PAYMENTS FOR ECOSYSTEM SERVICES IN A COUPLED HUMAN-NATURE SYSTEM

In collaboration with

Ashton Shortridge, Andrés Viña, and Jianguo Liu

ABSTRACT

Payments for ecosystem services (PES) have increasingly been implemented to protect and restore ecosystems worldwide. The efficiency of conservation investments in PES may differ under different policy arrangements. In addition, the effects of PES programs may be uncertain due to uncertainties in human responses to policies and complex depending on the dynamic human-natural interactions. To demonstrate the impacts of human-environment interactions on the effects of PES programs, we developed a spatially explicit model, human and natural interactions under policies (HANIP). We used HANIP to study the effects of China's Natural Forest Conservation Program (NFCP) and alternative policy scenarios in a coupled human-nature system (China's Wolong Nature Reserve for giant pandas), where indigenous people's use of fuelwood affected panda habitat. We estimated the effects of the current NFCP providing cash payment and an alternative payment scenario providing electricity payment by comparing habitat dynamics under these policies to habitat dynamics where no payment is provided. By 2030, there will be 107.68 km^2 of panda habitat in the study area if no payment is provided. Under the current NFCP, about 31.9% of habitat area can be obtained from the conservation payment by 2030. If the cash payment is replaced with electricity payment, an additional 11.6% of habitat area can be obtained. In addition, the recovery rate of panda habitat will be decreasing due to increasing population and households. Conservation effects of the NFCP may be threatened by the behavior of newly formed households if they are not included in the payment scheme. Our study demonstrated the advantages of integrating dynamics in human activities with the natural environment. Our modeling

framework may also be applied to understanding the effects of conservation policies in other coupled human-nature systems.

5.1 INTRODUCTION

Humans have substantial and growing impacts on the Earth's ecosystems (Millennium Ecosystem Assessment 2005). For instance, humans have transformed between one-third and one-half of the land surface (Vitousek et al. 1997), resulting in biodiversity loss and ecosystem degradation worldwide (Wackernagel et al. 2002, Luck et al. 2004). Human alteration of earth is not limited to human-dominated areas, but is also common in many protected areas in the world (Liu et al. 2001, Curran et al. 2004). To counter this trend, conservation efforts, including payments for ecosystem services (PES), have been invested by governments, private sectors, and conservation non-government organizations (OECD 1997, Ferraro and Kiss 2002). Much of these efforts have been aimed at reducing human impacts through shaping human activities (Smith 1995, Zbinden and Lee 2005, Wunder 2008). To improve the efficiency of conservation investments, PES programs have been implemented to provide incentives directly to ecosystem services providers (Ferraro and Kiss 2002, Wunder 2007). However, the effects of PES programs depend on the program's design and dynamic interactions among components in coupled human and natural systems.

Coupled human and natural systems (CHANS) are integrated human and natural systems in which human components interact with natural components (Liu et al. 2007a, Liu et al. 2007b). Although the importance of human-nature interactions has been recognized for the sustainability of both human and natural systems (Foley et al. 2005, Millennium Ecosystem Assessment 2005), complex processes in these interactions and patterns emerged from such interactions have not been well understood. Lack of studies of CHANS is mainly because social and ecological sciences have traditionally been developed separately (Rosa and Dietz 1998). Understanding of CHANS, however, relies on the integration of both social science studies and natural science studies.

Traditional methods for human activities use household survey data to study the impacts of individual characteristics on behavior (Chen et al. 2009a). But little is known how dynamics of individual characteristics will interact to result in macro-level ecological dynamics. Moreover, these methods usually cannot address heterogeneities in human activities across landscapes. For instance, two land parcels in a landscape with similar biophysical features may have different dynamic trajectories due to heterogeneities in human activities. Studies of natural systems often use aggregated data to detect patterns and changes (Liu et al. 2001, Viña et al. 2007). However, it is difficult to understand driving processes of dynamic natural systems using aggregated ecological data because decision-making of humans is often at a lower level (e.g., person or household).

Originating in artificial intelligence and paralleling individual-based modeling in ecology, agent-based modeling (ABM) is a bottom-up method that simulates actions of individual "agents" (e.g., persons or households) and their interactions with the environment to produce the aggregated macro-level patterns and processes (Parker et al. 2003, An et al. 2005). Agents have autonomous actions and are capable of interacting with other agents. Because of these features, ABM has been successfully applied in ecological studies, such as those for land use/cover changes, to understand driving processes of environmental changes and explore plausible future trajectories and policy implications (Deadman et al. 2004, Manson and Evans 2007, Matthews et al. 2007). ABM is also an ideal tool for understanding responses of human activities to institutional transitions and the resulting macro-level environmental and social changes.

The development of an agent-based model is often on the basis of object-oriented programming in computer science. In object-oriented programming, each modular unit is an object that has its own state (represented by its attributes) and behavior (implemented by its methods). Implementations of objects are modules of the program, and are relatively separated among each other to reduce programming complexity whereas objects may change state and behavior in response to states and behavior of other objects. Due to similarity in the paradigm, both object-oriented programming languages (e.g., Java, C++) and tools (e.g., NetLogo, Swarm) have been used for developing agent-based models. In studies of human-natural interactions and responses to policies, ABM is usually parameterized with studies from traditional approaches that address factors affecting individual decision-making and forces driving environmental change.

To demonstrate the impacts of human-natural interactions on the effects of PES programs, we developed a spatially explicit model, Human and Natural Interactions under Policies (HANIP). We used HANIP to study the effects of China's Natural Forest Conservation Program (NFCP, one of the largest PES programs in the world) and alternative policy scenarios in Wolong Nature Reserve for providing habitat to giant pandas (*Ailuropoda melanoleuca*). HANIP incorporates agent-based modeling techniques for modeling dynamics in population, households, and their activities and regression-based models for corresponding changes in the landscape. We chose household as the unit of human activities because household is often the basic socioeconomic unit for land use decision-making and consumption of resources (Liu et al. 2003a, Manson and Evans 2007). We focused on fuelwood use and development of farmland for newly formed households because they are main forces of dynamics in forests. Changes in the quantity and distribution of forests result in changes in panda habitat. In HANIP, there are two major types of agents, person and household. Dynamics in these agents result in dynamic human impacts (fuelwood use and farming), which produce changes in panda habitat. HANIP was developed using Java programming language (JDK 1.4.2, Sun Microsystems).

5.2 METHODS

5.2.1 Model summary

The conceptual framework of HANIP is described in Figure 5.1. This framework was implemented in three submodels. In the demographic submodel, dynamics in people including population and households were modeled by simulating individual persons' life histories. Young adults may form new households after they get married, and newly formed households were geographically distributed around their parental households depending on the topographic conditions. Households' fuelwood use was modeled in the policy submodel (Figure 5.1). Under different policy scenarios, each household would use different amounts of fuelwood, which was also determined by household characteristics and topographic conditions. Both households' fuelwood use and increased farming of new households affect dynamics in forest cover and panda habitat that were modeled in the landscape submodel (Figure 5.1). Changes in forest cover also depend on topographic conditions. Since forest cover is one of the most important components of panda habitat that is also constrained by topographic conditions, dynamics in forest cover may result in changes in panda habitat.

5.2.2 Demographic submodel

We obtained characteristics of all households in the reserve and their corresponding household members, including age, gender, kinship relation, marital status of household members and amount of cropland of households that are all available in three possible sources, the 1996 agricultural census (4053 residents in 892 households), the 2000 population census, (4375 residents in 969 households) and the 2006 household registration (4505 residents in 1197 households). The geographic locations and elevations of all households were measured using Global Positioning System (GPS) receivers.

Population and household dynamics were studied in the reserve (An et al. 2001, An et al. 2003). Population dynamics was modeled by simulating individual persons' life histories in one year increments. Major events of person agents included married female give birth, students move out of the reserve through going to college, single persons get married, people move in or out of the reserve through marriage, grow by one year and die. These events were modeled as stochastic processes based on person agents' state that included age, gender, kinship relation, and marital status (An et al. 2001). The state of household agents included location and elevation of households, amount of cropland, household size and availability of senior people

(>60 years old). When young adults get married, a new household may be formed, which was set as a stochastic process and depended on the gender of the young adult, whether the young adult has siblings, the age of the young adult compared to siblings, and the young adult's intention of forming a new household (An et al. 2003).

Our demographic submodel adopted previous findings in the reserve (An et al. 2001, An et al. 2003) with the exception that we updated the young adults' probability of moving out of the reserve through going to college using the 2006 household registration data. On the basis of previous studies in this area, farmlands were associated with their corresponding households. Since the average farmland area of households was only 0.28 ha (Chen et al. 2009b), farmland and buildings of each household were located in the same pixel (90 m by 90 m). Newly formed households were stochastically located on areas with slopes < 37 degrees and within 800m from their corresponding parental households; these parameters were based on information from existing households (An et al. 2005). We assumed that farmland is divided proportionally to household size when a new household is formed. Major events of household agents also included change in household size depends on the life history of each household member and dissolution of a household when there is no member in the household.

5.2.3 Policy submodel

We designed three policy scenarios: no payment, cash payment under the current NFCP, and electricity payment that substitutes cash payment at the same cost. Under the no payment scenario, households' fuelwood use was assumed to follow the fuelwood use pattern prior to the NFCP. Households' fuelwood use pattern prior to the NFCP was modeled on the basis of household size, availability of senior people in the household, and farmland area (An et al. 2001). On average, each household used 15 m^3 of fuelwood per year prior to the NFCP. To understand households' fuelwood use patterns under the cash payment and electricity payment, we randomly chose 321 households from a total of 1197 households for in-person interviews (305 valid interviews, 95% response rate) in the summer of 2006. We chose household heads or their spouses as interviewees because they are usually the decision-makers of household affairs. We asked the average amount of daily fuelwood consumption in the previous year for both the winter season when more fuelwood is consumed and the summer season when less fuelwood is consumed. Household fuelwood use was measured as a summation of daily consumption across the year. We also asked about the amount of fuelwood that each household would demand if the cash payment of the NFCP was substituted with electricity payment at the price in 2006 (0.18 yuan/kW*h), which leads to an electricity payment of about 14 kW*h per day.

Interviews with 305 households showed an average household fuelwood consumption of about 9 m^3 per year under the current NFCP with cash payment. Households' fuelwood use pattern under the cash payment and electricity payment was modeled on the basis of the household characteristics that are available to all the households in the reserve (Table 5.1). We corrected for the correlation of households' responses under the cash payment and the electricity payment using Huber's variance correction (Wooldridge 2002). Households' fuelwood use was significantly positively correlated to household size because more people in a household usually required more fuelwood for cooking and heating. Farmland area of households significantly positively correlated to households' fuelwood use because households with more cropland usually used more crops to feed more pigs, and fuelwood for cooking pig fodder was an important part of households' fuelwood use. Elevation of households also significantly positively correlated to fuelwood use because households living at a higher elevation usually need more fuelwood for heating in winters than those at lower elevations due to differences in microclimate.

If the cash payment was substituted with electricity payment, households' fuelwood use would be reduced by 3.1 m^3 per year on average (Table 5.1). Electricity payment was more efficient than cash payment in reducing fuelwood use because all the electricity payment would be used to replace fuelwood whereas not all the cash payment may be used for electricity. Compared to An et al. (2001), availability of senior people in the households was not significantly correlated to fuelwood use. This is probably because extra fuelwood use by senior people for heating was reduced under the NFCP. Although new households were not included in the NFCP, we did not find significant differences in fuelwood use between new households and other households partly because all parental households of new households were enrolled in the NFCP. As the number of new households increases, their fuelwood use pattern can be uncertain if they continue to be excluded from the NFCP.

5.2.4 Landscape submodel

Forest distributions in Wolong were generated from classification of remotely sensed imagery (Landsat Thematic Mapper) acquired on June 26, 1994, June 13, 2001 and September 18, 2007. We used an unsupervised classification based on the ISODATA technique, which is an iterative process for non-hierarchical pixel classification (Jensen 1996). A maximum of 1,000 iterations were used for classification, and produced an output of 100 spectral classes. We then applied a post-classification sorting method and merged the 100 spectral classes into four information classes: forest, non-forest, clouds and cloud shadows through a combination of visual interpretation of these images and information on land cover obtained from high spatial resolution multispectral imagery (i.e., four IKONOS multi-spectral scenes (4 x 4 m / pixel) acquired on August 31, October 3, and November 8 and 16 of 2000, respectively and a Quickbird multi-spectral scene (2.4 x 2.4 m / pixel) acquired on November 23, 2007). A few areas under cloud and cloud shadows were excluded from further analysis. The accuracy of classifications were assessed using ground truth points collected during the summers of 1998 (209 points), 2000 (83 points), 2001 (83 points) and 2007 (593 points) that were measured using real-time, differentially corrected GPS receivers. The overall accuracies of classified forest distributions were 79.2%, 78.2% and 82.6% for the 1994, 2001 and 2007 imagery, respectively.

From 1994 to 2001, there was 20.1% forest loss and 12.0% forest recovery, resulting in 8.6% net deforestation. From 2001 to 2007, there was 12.4% forest loss and 22.7% forest recovery, resulting in 10.3% net forest regeneration. Forest dynamics (i.e. forest loss and forest recovery) were analyzed at pixel level, and a 90 m by 90 m pixel size was chosen based on the availability of topographic data and computational complexity. We randomly selected 4500 pixels, where two-thirds of the data (3000 pixels) were used for model calibration and one-third of the data (1500 pixels) were used for model validation. Among the 3000 pixels for model calibration, 1982 and 1797 pixels were forest pixels in 1994 and 2001 respectively, pooled to a total of 3779 forest pixels, and 1018 and 1203 pixels were non-forest pixels in 1994 and 2001 respectively, pooled to a total of 2221 non-forest pixels. We then modeled forest loss in 1994-2001 or 2001-2007 with 3779 forest pixels and forest recovery in 1994-2001 or 2001-2007 with 2221 non-forest pixels using two logistic regression models (Table 5.2 and Table 5.3). We corrected for dependencies between pixels that represented both 1994-2001 and 2001-2007 periods with Huber's variance correction (Wooldridge 2002).

We used elevation, slope, aspect [converted into soil moisture classes (Parker 1982)] and distance to forest edge, that were used in previous studies of forest dynamics (Geoghegan et al. 2001, Nagendra et al. 2003). In addition, we used a inverse distance weighted fuelwood impact variable that was measured as the aggregation of the impacts of all eligible households (households within 6-km buffer from the pixel, Table 5.2 and Table 5.3) on each pixel. The impact of each eligible household on a pixel was measured as the household's fuelwood consumption divided by the distance between the household and the pixel. We also used total fuelwood use of all households as an explanatory variable. Annual fuelwood use of households for forest dynamics between 1994 and 2001 were estimated using household characteristics from 1996 agricultural census data and fuelwood model developed prior to the NFCP (An et al. 2001), while annual fuelwood use of households for forest dynamics between 2001 and 2007 were estimated using household characteristics from 2006 registration data and fuelwood model under the

NFCP that was developed in the previous section. We did not find multicollinearity among independent variables.

Although the duration of forest dynamics between 1994 and 2001 was 7 years, the NFCP enrollment took place in 2000. Therefore we assumed the deforestation trend before the NFCP was due to fuelwood use in the first 6 years. The duration of forest dynamics between 2001 and 2007 was also 6 years. Since HANIP was built on a yearly basis, we approximated the annual dynamics in each pixel by dividing the estimated probabilities of forest loss and forest recovery from these models by 6 years. Although topographic variables (elevation, slope, and aspect) of pixels do not change over time in HANIP, distance to forest edge changes in each year as forest cover changes. In addition, both fuelwood impact and total fuelwood change in each year depending on changes in population and households. We estimated probability of forest loss for forest pixels and probability of forest recovery for non-forest pixels in each year in HANIP, and determined annual forest dynamics of each pixel with a Bernoulli trial. The rate parameters of the Bernoulli distributions were the probability of forest loss or forest recovery.

Pixels may be classified as different levels of panda habitat suitability based on forest cover and topographic factors. Pixels with forest cover, an elevation between 1,500 and 3,250 m, and a slope less than 30 degrees were classified as highly suitable habitat and suitable habitat by Liu et al. (1999), and were combined as habitat in our model. In HANIP, forest loss in panda habitat due to fuelwood collection and farming resulted in the loss of panda habitat, while forest recovery in areas with suitable topographic conditions led to the recovery of panda habitat. Comparisons of habitat quantity under different policy scenarios allowed detection of the impacts of conservation investments.

5.2.5 Model validation

Our simulation model was run 30 times under each set of parameters to obtain results of stochastic processes. Due to the limitation on the data availability, the demographic model and the landscape model started in different years. For model validation, we used 2000 population census data as the starting point for the demographic model and 2001 land-cover data as the starting point for the landscape model. Our model validation included comparison of simulation results with empirical data and sensitivity analysis. We evaluated the landscape submodel by testing regression models for forest loss and forest recovery using a receiver operating characteristic (ROC) curve (Hanley and Mcneil 1982). The ROC curve is a plot of the sensitivity values (i.e., true positive fraction) vs. their equivalent 1-specificity values (i.e., false positive fraction) for all possible probability thresholds. The area under the ROC curve (AUC) is a measure of model accuracy, with AUC values ranging from 0 to 1, where a score of 1 indicates perfect discrimination, a score of 0.5 implies a prediction that is not better than random, and lower than 0.5 implies a worse than random prediction. We used the validation data set (i.e. one-third of the 4500 randomly selected pixels) for deriving the AUC value. We also compared the observed habitat area in 2007 to the mean of predicted habitat areas in 2007 from 30 simulation runs.

For demographic submodel, we compared the observed population size and number of households in 2006 to the simulation results. Although the validation of the effects of policy scenarios on panda habitat was not feasible, measurement of the impact of the current cash payment relied on the validation of the landscape submodel, and the fuelwood use pattern under the circumstance where no payment was provided and validated in a previous study (An et al. 2001). Finally we conducted sensitivity analysis to evaluate how sensitive model results were to small changes in several key model parameters (Haefner 1997). The sensitivity index is defined as Sx = $(\Delta Y/Y_0)/(\Delta X/X_0)$, where X_0 is the initial value of a model parameter, ΔX is a small change in X, Y_0 is the initial outcome, and ΔY is the corresponding change in Y due to the change in X. Small sensitivity values, suggesting robustness of the outcome to small changes in parameters, are usually preferred. We also used two-sample t-test to examine differences in simulation results due to the changes in these parameters.

5.2.6 Simulation experiments

We used 2006 household registration data as the starting point for the demographic model and 2007 land-cover data as the starting point for the landscape model and run simulations through 2030. To demonstrate the conservation effects of the NFCP and the electricity payment scenario, we also used households' fuelwood use pattern prior to the NFCP (An et al. 2001) in the circumstance where no payments were provided. Since new households are currently not included in the NFCP, their fuelwood use pattern is uncertain as the number of new households increases in the future. To explore this uncertainty, we also predicted habitat area under circumstances where half and all of new households follow the fuelwood use pattern prior to the NFCP.

5.3 RESULTS

5.3.1 Model validation

Although accurate prediction of forest dynamics at the pixel level is difficult, both forest loss and forest recovery models exhibited moderately high accuracy, with AUC values of 0.775 and 0.773 respectively. Comparisons of model predictions and observed values showed that predicted mean habitat area in 2007 was 122.77 km^2, which was close to the observed value (Table 5.4). The difference between the mean predicted habitat area and the observed habitat area was 0.39 km^2, which was less than the observed mean yearly change in habitat (2.02 km^2) from 2001 to 2007. The difference between the predicted mean human population and observed human population in 2006 was 12, which was also less than the mean yearly population change (22) from 2000 to 2006. The observed number of households in 2006 was 1197, which were 60 more than the predicted mean households. This difference was mainly because an unexpectedly large number of new households were formed in 2001, following the implementation of the NFCP, to more effectively capture conservation subsidies that are distributed on the basis of household (Liu et al. 2007a).

Habitat area was insensitive to small changes in all four selected parameters in sensitivity analyses (Table 5.5). A 10% increase in juveniles' college entrance rate resulted in 0.019 km^2 increase in mean habitat (Sx = 0.002), while a 10% increase in young adult's intention of forming a new household resulted in 0.063 km^2 decrease in mean habitat (Sx = -0.005). A 50% (400 m) increase in the maximum distance between a newly formed household and its parental household decreased mean habitat by 0.034 km^2 (Sx = -0.001). Statistical tests showed that perturbations in these parameters did not result in significant differences in mean habitat area (Table 5.5). A 10% decrease in the effect of electricity payment (average amount of fuelwood that can be saved by replacing cash payment with electricity payment) significantly decreased mean habitat by 0.384 km^2 ($p = 0.001$), even though habitat area was insensitive to the effect of electricity payment (Sx = -0.030). Significant decrease in mean habitat area due to decrease in the effect of electricity payment was expected because change in fuelwood was directly related to habitat dynamics in our model.

5.3.2 Simulation experiments

Compared to panda habitat and households in 2007 (Figure 5.2), both panda habitat and households varied under no-payment scenario (Figure 5.3), cash payment scenario (Figure 5.4) and electricity payment scenario (Figure 5.5). Under the cash payment of the NFCP, habitat area will be increased from 123.16 km^2 in 2007 to 142.07 km^2 in 2030, corresponding to an average yearly increase of 0.82 km^2 (Figure 5.6). If cash payment is replaced with electricity payment, the average yearly increase will be 1.36 km^2, resulting in a total habitat area of 154.49 km^2 in 2030. Compared to the fuelwood use pattern prior to the NFCP under the no-payment scenario, the effects of cash payment and electricity payment are increasing (Figure 5.6). By 2020, 21.10 km^2 (18.3%) and 29.52 km^2 (25.5%) of habitat area can be gained, and by 2030, 34.38 km^2 (31.9%) and 46.81 km^2 (43.5%) of habitat area can be gained through cash payment and electricity payment, respectively. The increase in habitat area is non-linear. From 2011 to 2015, the average yearly increase in habitat area is 1.09 km^2

under the cash payment, while the average yearly increase rate reduced to 0.44 km^2 from 2025 to 2030. This non-linearity is because of increases in population and households. In 2015, there will be about 4720 people in about 1340 households in the reserve, and the population and households will be increased to about 4950 and 1460 by 2030.

Dynamics in habitat area will also depend on the behavior of newly formed households (households formed after 2001). The more the new households follow the fuelwood use pattern prior to the NFCP, the less panda habitat will be gained from the conservation payment (Figure 5.7). Compared to the fuelwood use pattern under the no-payment scenario (Figure 5.6), 29.79 km^2 (27.7%) and 24.87 km^2 (23.1%) of habitat area can be gained if half and all of new households follow the fuelwood use pattern prior to the NFCP, which are not much less than if all households follow the current fuelwood use pattern (Figure 5.7). However, habitat area will start decreasing in 2028 if all of new households follow the fuelwood use pattern prior to the NFCP.

5.4 DISCUSSION

The trend of dynamics in forest cover in Wolong Nature Reserve has changed from rapid deforestation between 1994 and 2001 to rapid forest regeneration between 2001 and 2007. Although such an abrupt change could be contributed by multiple factors, the Natural Forest Conservation Program (NFCP) dramatically reduced local households' fuelwood use through increased affordability to electricity use with conservation payment. While the NFCP has been successful in conserving habitat of giant pandas and many other wildlife species, its efficiency may be improved under alternative policy arrangements. In addition, there are uncertainties in the effects of such conservation investments due to complex interactions between human and the environment (Liu et al. 2007a).

We developed a spatially explicit model, HANIP, to study human and natural interactions under policies. Simulation experiments using HANIP under different policy scenarios allowed us to measure the conservation effects of different policies. By 2030, 31.9% and 43.5% of panda habitat in the study area can be obtained from cash payment and electricity payment, respectively. Compared to cash payment, conservation payment in the form of electricity is a more direct approach of paying people to reduce their negative impacts by replacing fuelwood with electricity. Therefore, electricity payment may improve the efficiency of conservation investments. As current conservation investments are far below the requirements for conserving ecosystems globally (James et al. 1999, James et al. 2001), it is important to improve the efficiency of existing conservation investments. We recognized that there are potential negative environmental impacts from electricity generation, which should also be considered for policy implementation.

Through modeling dynamics in population and households, HANIP can also detect changes in human impacts and policy effects across time. Non-linear increases in habitat area under the current payment scheme suggested that conservation gains from PES programs may largely depend on the dynamics in indigenous communities, such as dynamics in households and population. In addition, there are uncertainties in conservation gains due to uncertain human responses to policy arrangements. In our case, the effect of conservation payment may be threatened by the behavior of newly formed households if they are not included in the program. Uncertainty may also exist in the part where not all recovered forest areas, constrained by topographic conditions, may be used by giant pandas immediately after forest recovery. Future studies may explore the lagged effects of forest recovery on panda habitat.

Interactions among components in coupled human and natural systems (CHANS) are complex (Liu et al. 2007a). However, study of these complex interactions is important for understanding the effects of conservation investments that may involve complexity (e.g., non-linearity) and uncertainty. By modeling some of the key dynamic interactions, HANIP can provide important implication to existing PES programs, such as the NFCP, and conservation investments in the future. The modeling framework of HANIP may also be used to study conservation policies in other CHANS.

Table 5.1. Pooled Ordinary Least Squares of fuelwood use pattern.

Independent variables	**Parameters**	**Robust SE**
Household size	0.441*	0.241
Farmland (ha)	5.742**	2.522
Elevation (100 m)	0.746****	0.154
Electricity payment (dummy)	-3.097****	0.267
Constant	-8.164***	3.074
R-squared	0.12	

Dependent variable: fuelwood use (m^3). Observations: 610.
Significance: * $p \leq 0.1$; ** $p \leq 0.05$; *** $p \leq 0.01$; **** $p \leq 0.001$.

Table 5.2. Pooled logit estimation of forest loss.

Independent variables	Parameters	Robust SE
Elevation (100 m)	-0.008	0.014
Slope (degree)	0.001	0.006
Aspect (Parker scale)	-0.054***	0.008
Distance to forest edge (m)	-0.019***	0.001
Fuelwood impact (m^3/m)	0.031***	0.008
Total fuelwood (1000 m^3)	0.014***	0.002
Constant	-1.279**	0.491
χ^2	347.46***	

Observations: 3779.

Significance: ** $p \leq 0.01$; *** $p \leq 0.001$.

Table 5.3. Pooled logit estimation of forest recovery.

Independent variables	Parameters	Robust SE
Elevation	-0.008	0.011
Slope	-0.009	0.006
Aspect	0.064***	0.010
Distance to forest edge	-0.014***	0.001
Fuelwood impact	-0.009	0.008
Total fuelwood	-0.016***	0.002
Constant	1.352***	0.384
χ^2	263.71***	

Observations: 2221.

Significance: *** $p \leq 0.001$.

Table 5.4. Comparison of model predictions of panda habitat, population size and household number to observed values.

Factors	Observed value	Observed mean yearly change	Model mean	Difference between model mean and observed value	|Difference| < observed mean yearly change
Habitat in 2007 (km^2)	123.16	2.02	122.77	-0.39	Yes
Population in 2006	4505	22	4493	-12	Yes
Households in 2006	1197	38	1137	-60	No

Table 5.5. Sensitivity tests for selected model parameters.

Parameters	Default value	Perturbation	Change in habitat area (km^2)	t statistic for differences in habitat area (p-value)	Sensitivity
College entrance rate	0.274	+0.0274 (10%)	0.019	0.169 (0.867)	0.002
Separate home intention	0.42	+0.042 (10%)	-0.063	-0.841 (0.407)	-0.005
New household location	800 m	+400 (50%)	-0.034	-0.381 (0.706)	-0.001
Electricity payment effect	3.1 m^3	-0.31 (10%)	-0.384	-3.534 (0.001)	-0.030

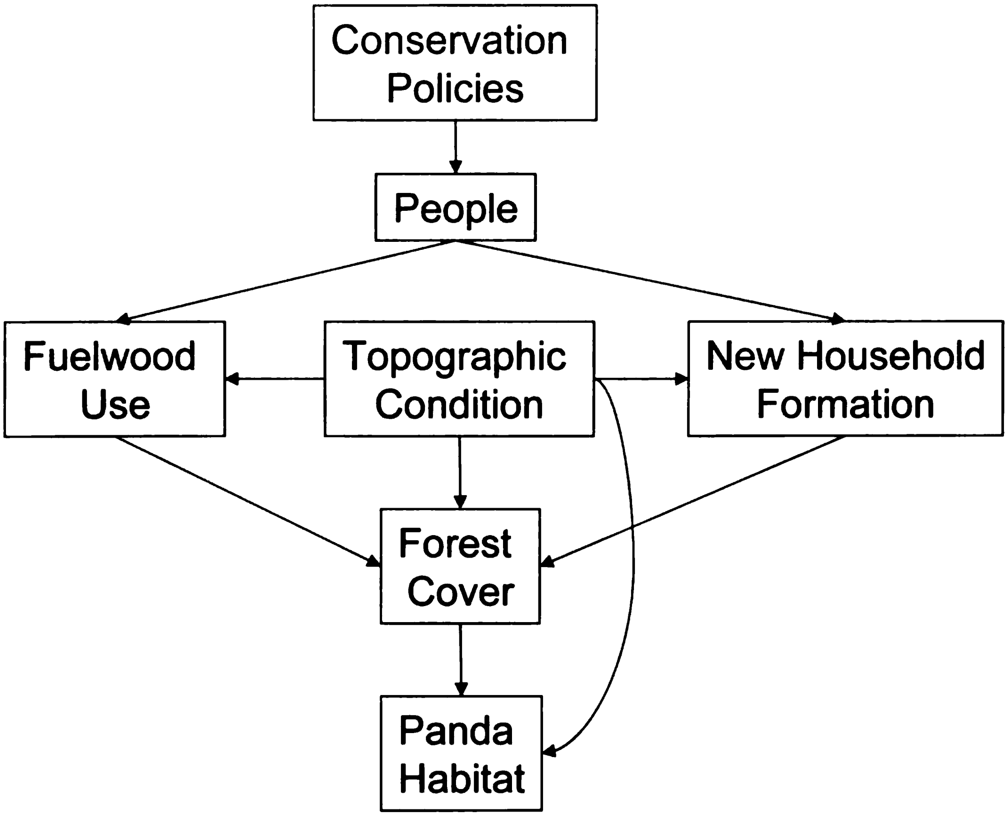

Figure 5.1. Conceptual framework of the model.

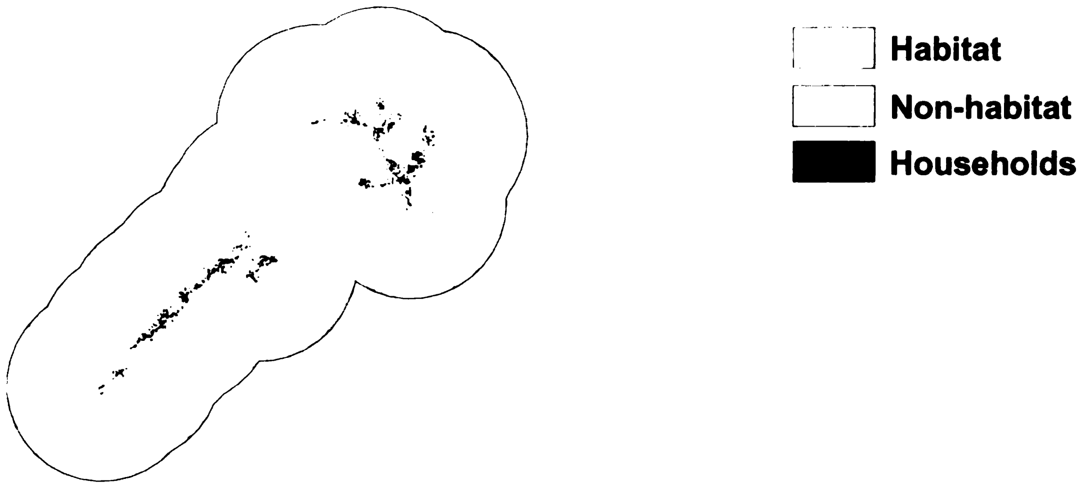

Figure 5.2. Panda habitat and household distribution in 2007.

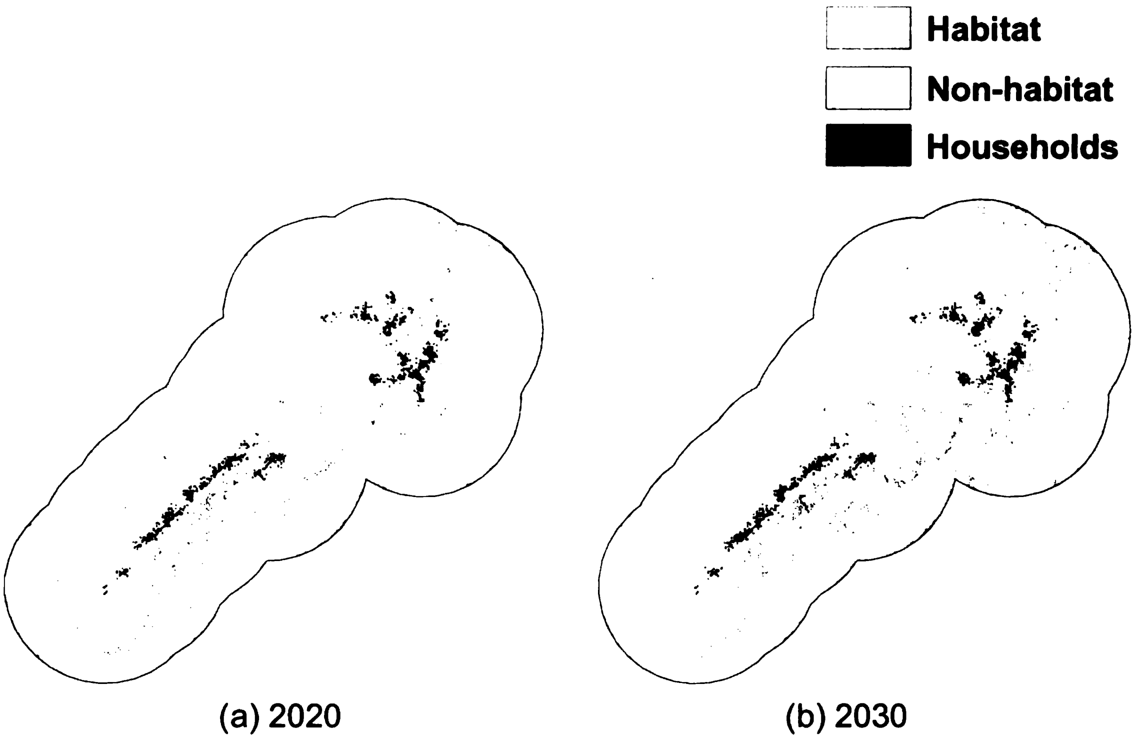

Figure 5.3. Panda habitat and household distribution in 2020 (a) and 2030 (b) under no-payment scenario from one run.

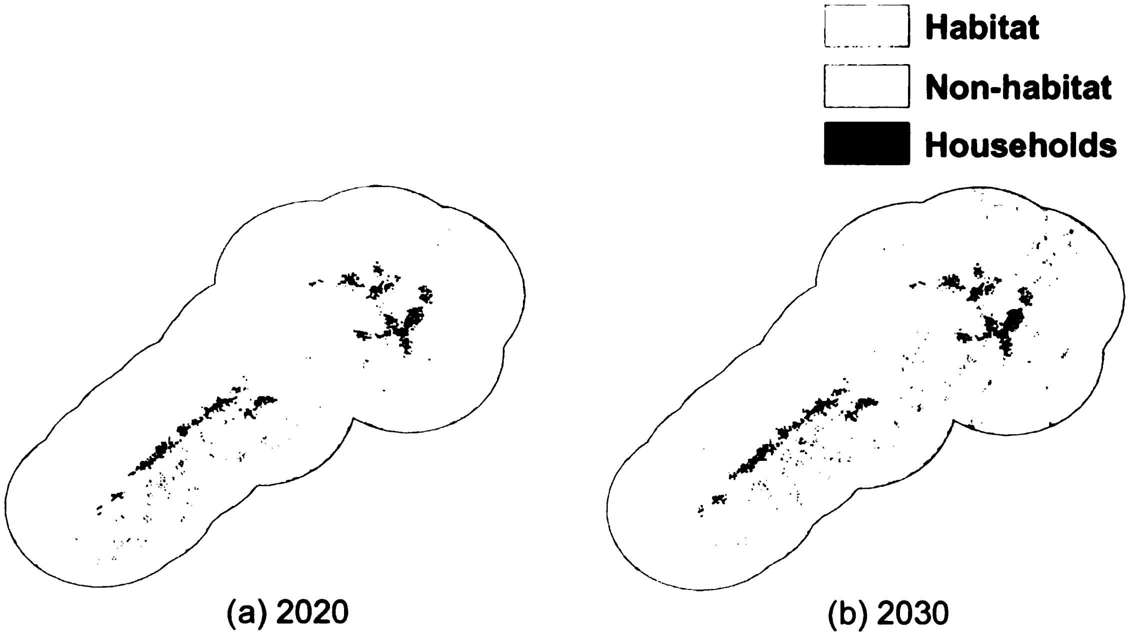

Figure 5.4. Panda habitat and household distribution in 2020 (a) and 2030 (b) under cash payment scenario from one run.

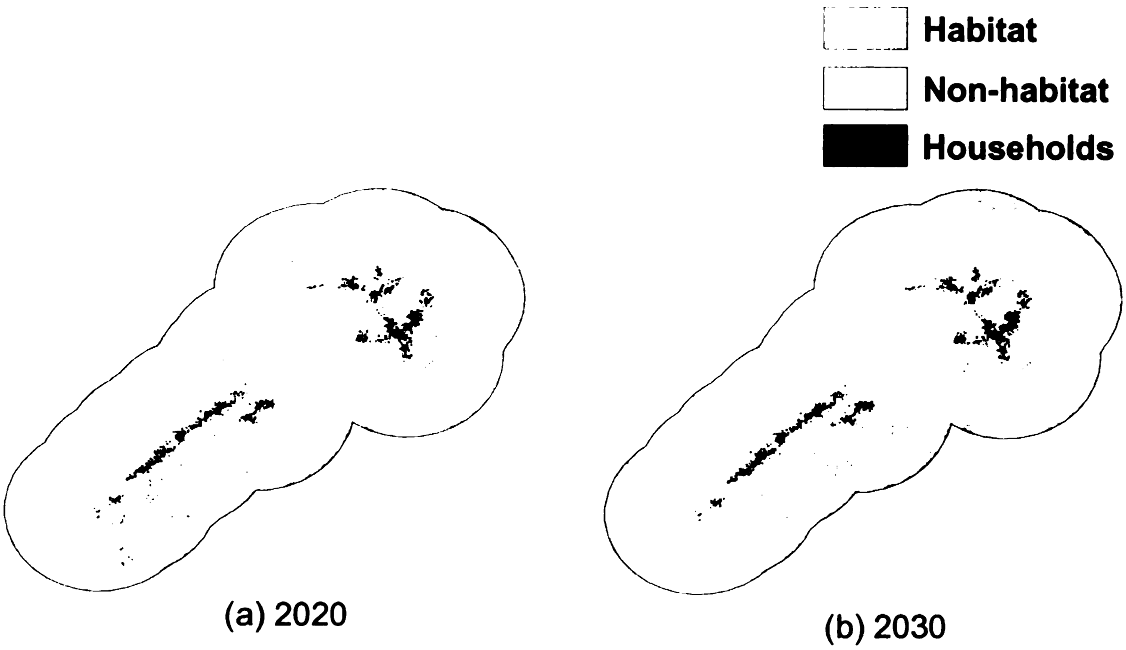

Figure 5.5. Panda habitat and household distribution in 2020 (a) and 2030 (b) under electricity payment scenario from one run.

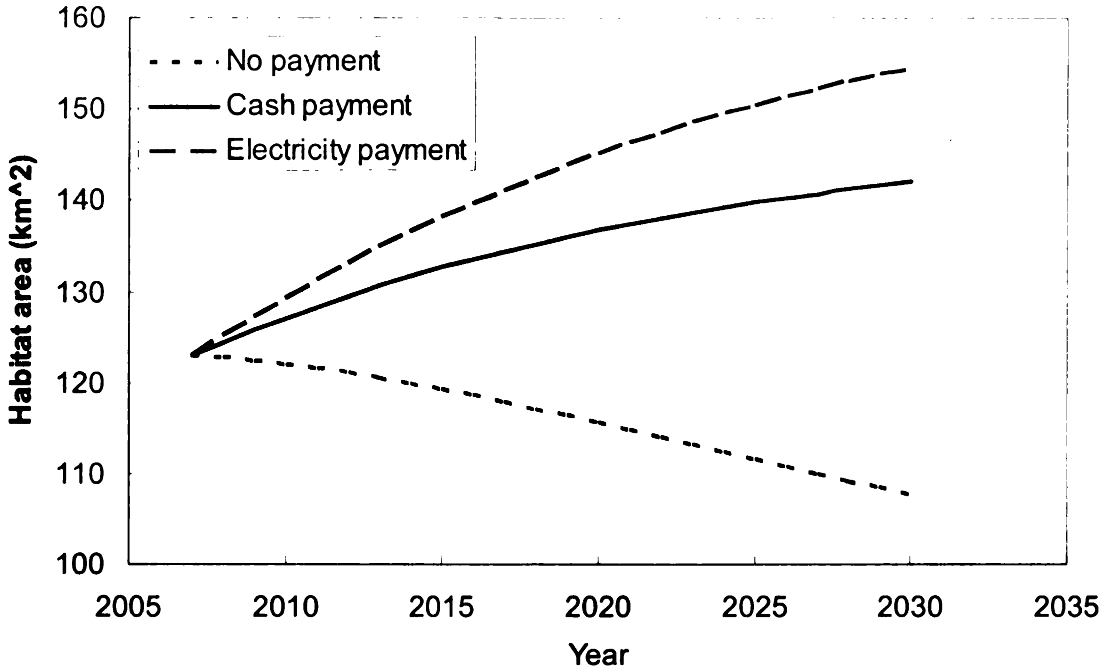

Figure 5.6. Predicted panda habitat under cash payment, electricity payment, and no payment scenarios. We did not draw confidence intervals because standard deviations from 30 runs are small.

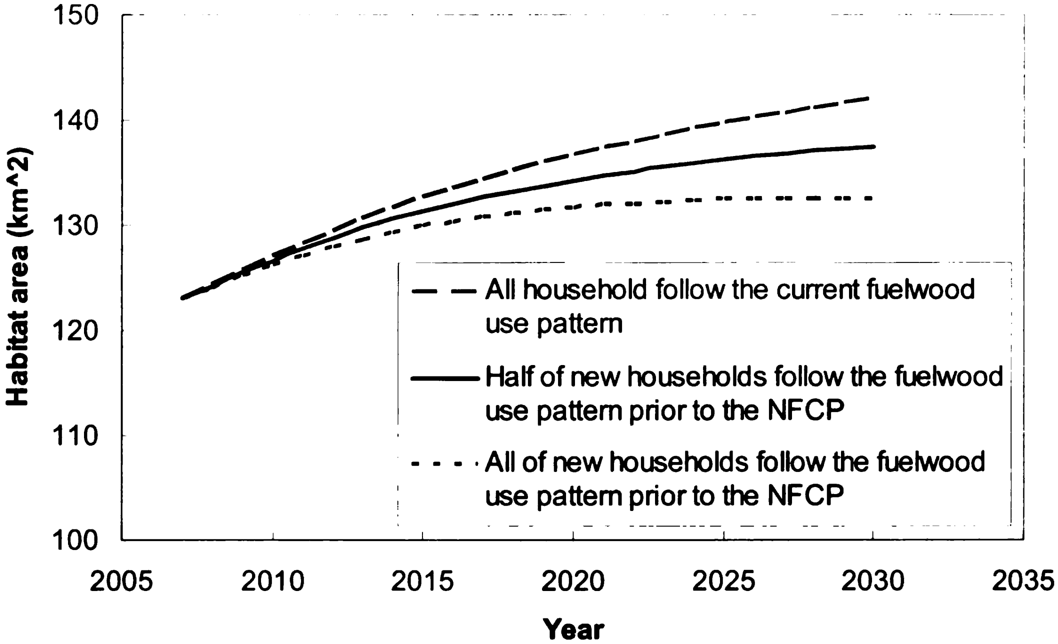

Figure 5.7. Predicted panda habitat under different fuelwood use patterns followed by new households (households formed after 2001).

CHAPTER 6

CONCLUSIONS

Findings from this dissertation provide important implications to the management of local environment in Wolong Nature Reserve, China's current and future conservation efforts, and global practices of conservation investments through PES. Using a stated-choice model with a main effects design, we found that social norms had substantial effects on participants' intentions of re-enrolling their GTGP land, suggesting people's enrollment intentions tend to conform to the majority (Chapter 2). The extra cost for obtaining an additional unit of land for conservation would be lower if most people in a community would enroll their land in a PES program. In addition, increased conservation investments can leverage social norms in communities where most people wound initially not participate. The aggregated impacts of social norms can be substantial.

This study contributed to the literature by suggesting that social norms should be integrated with biological values, economic conditions and demographic trends for efficient conservation investments. In addition, stated-choice model using a main effects design can relatively easily distinguish the effects of social norms from the effects of other factors, which was a challenge in the past studies on social norms (Manski 2000). The effects of social norms can also be substantial in many other environmental issues, such as common-pool resources management and marketing for environmentally friendly business. The approach and methods in this study can also be applied when addressing those environmental issues.

We also found that different sources of income had different effects on program re-enrollment (Chapter 2). Farming income had a negative effect on program re-enrollment; however, income from rural-urban labor migrants had a positive effect on program re-enrollment. Although off-farm income may lower farmers' dependence on the lands that are enrolled in PES programs, off-farm income from employment within the reserve did not have such an effect. Compared to off-farm employment outside of the reserve, off-farm employment within the reserve is much more flexible in terms of labor and time allocation. These results suggested that not all off-farm income may increase land enrollment in PES programs, and different types of off-farm employment should be treated differently. The trend of rural-urban labor migration in transitional economies of many developing countries (United Nations 2004, Korinek et al. 2005) provides a great opportunity for PES programs to lower costs and sustain the gains from these investments.

Currently conservation investments through PES have been implemented in many countries. The design and implementation of these PES programs are different. For instance, both flat payments (all participants paid the same price) and discriminative payments (participants paid different prices according to opportunity costs) have been used in PES programs (Claassen et al. 2008, Pagiola 2008), resulting in different levels of efficiency in conservation investments. For the GTGP, there are only two payment levels nationwide which operate as flat payments within each region. To demonstrate the potential of improving the efficiency of conservation investments in the GTGP, we cost-effectively targeted land for maximizing environmental benefits obtained from the GTGP (Chapter 3). We used environmental benefits of and cost for lands that were estimated on the basis of land features and household characteristics. The results suggested that the efficiency of the GTGP can be improved up to ten times by switching from the flat payment scheme to cost-effective targeting in a discriminative payment scheme.

This study made another contribution to the literature by suggesting household characteristics and regional differences as significant determinants of opportunity costs of landholders participating in the GTGP (Chapter 3). Therefore, household characteristics and regional differences should be incorporated with biological values and physical conditions of lands in the planning of PES programs. Our findings highlighted the importance of integrating human systems with natural systems for efficient conservation policy design. In the practice of conservation investments, however, opportunity costs and many household characteristics of landholders are often not available to the public. Competitive auctions have successfully been applied in some PES programs. Cost-effective targeting coupled with competitive auctions could greatly improve the efficiency of conservation investments through PES and other conservation policies.

In addition to its direct conservation gains through conversion of agricultural land to natural vegetation cover, studies on the GTGP suggested that the GTGP has boosted the trend of rural-urban labor migration by releasing many laborers from agriculture (Bao et al. 2005, Liu 2005, Ge et al. 2006, Hu et al. 2006, Uchida et al. 2009). While formal institutions facilitating rural-urban labor migrants have been rare in rural regions in China and many other countries, social capital is an important source providing employment information in urban regions (Granovetter 1995, Bian 1997) and facilitating migration processes (Massey and Espinosa 1997, Korinek et al. 2005). We studied the effects of labor migration on fuelwood consumption from the perspective of capital substitution (Chapter 4). Social capital, especially weak social ties, was often used to gain off-farm income through labor migration. Off-farm income from labor migration then displaced the use of fuelwood (a form of natural capital) through increased affordability to electricity. Additionally, reduced population pressure on the local ecosystem and reduced labor supply for fuelwood collection due to labor migration also resulted in the decreases in fuelwood consumption.

People in many rural and ecologically significant regions, such as in our study area, usually have limited access to employment information in urban regions. This study also contributed to the literature by linking social capital with natural capital through labor migration. These results suggested policy instruments that aim at reducing human impacts in ecologically significant regions should provide employment information and facilitate labor migration (Chapter 4). Numerous off-farm employment opportunities produced from transitional economies in many developing countries can be a great opportunity for conservation actions in these countries.

Even with substantial conservation efforts (e.g., investments), ecosystem conservation in CHANS presents a formidable challenge. This is at least partly due to complexity that is inherent to CHANS (Liu et al. 2007a). For instance, people tend to think linearly, while the effects of conservation investments can be non-linear due to complex human-natural interactions. In addition, people may respond to conservation policies differently under different circumstances, resulting in uncertainty in the effects of conservation policies. Systems models can be useful tools for understanding complexity, such as uncertainty and non-linearity, in many CHANS.

The systems model that was developed for this dissertation (Chapter 5) quantified the effects of the Natural Forest Conservation program (NFCP) and alternative policy scenarios on panda habitat. Compared to the non-payment scenario, substantial gains in panda habitat area can be obtained from both cash payment under the current NFCP and the electricity payment scenario. Electricity payment, as a more direct payment approach for reducing human impacts on panda habitat, can improve the efficiency of conservation investments. The conservation gains from conservation payments will decrease as both human population and number of households increase. Moreover, the effects of conservation payments also depend on the behavior of newly formed households (households that have been formed after 2000) because these new households have not been included in the NFCP. Conservation gains from the NFCP and the electricity payment scheme can be uncertain due to uncertainties in the behavior of new households.

These outcomes highlighted the importance of integrating dynamic human society with the changing natural environment in CHANS. By integrating dynamic human population, households, and activities with land use changes using systems models, long-term effects of the NFCP can be detected for policy evaluation. Comparisons among different policy scenarios provide important support to decision-making of governments and conservation practitioners. Our modeling framework could also be used in other CHANS especially for evaluating conservation investments through PES.

This dissertation focused on the efficiency and effectiveness of payments for ecosystem services in China's Wolong Nature Reserve. Because the objectives in this research involve both the indigenous community and the natural environment, we used interdisciplinary methods and tools. Interesting findings have been produced through this approach. Future research on the efficiency and effectiveness of PES, such as how pro-environmental social norms can be formed through PES, understanding the dynamic impacts of social norms and social capital (e.g., weak ties) using systems models, is needed.

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