Anthropocene 31 (2020) 100255

Contents lists available at ScienceDirect

Anthropocene

journal homepage: www.elsevier.com/locate/ancene

Understanding direct and indirect effects of Payment for Ecosystem Services on resource use and wildlife

Hsiang Ling Chen[a,*], Rebecca L. Lewison[b], Li An[c], Shuang Yang[c], Lei Shi[d], Weiyong Zhang[d]

[a] Department of Forestry, National Chung Hsing University, 145, Xingda Rd., Taichung, 402, Taiwan
[b] Department of Biology, San Diego State University, 5500 Campanile Drive, San Diego, CA 92182-4614, USA
[c] Department of Geography, San Diego State University, 5500 Campanile Drive, San Diego, CA 92182-4493, USA
[d] Fanjingshan National Nature Reserve Administration, 17 Sanxing West Rd., Shuangjiang Township, Jiangkou County, Guizhou Province, China

ARTICLE INFO

Article history:
Received 31 August 2019
Received in revised form 20 July 2020
Accepted 21 July 2020
Available online 24 July 2020

Keywords:
Grain-to-Green program
Human activity
Guizhou
Coupled human and natural system
Multi-species hierarchical occupancy models
Forest management

ABSTRACT

Payments for ecosystem services (PES) programs have been implemented worldwide to balance human needs and ecosystem conservation. However, the effects of PES programs on economic activities and the associated effects on wildlife remain unclear. China's Grain-to-Green program (GTGP) that aims to convert cropland to forest or grassland represents one of the largest PES programs in the world. We combine household surveys with wildlife camera data to ask whether GTGP is associated with direct or indirect effects on economic activities, wildlife occupancy and wildlife species richness. We also consider how resource use and extraction, wildlife occupancy, and species richness differ between GTGP and natural forest sites. We found that farming and the amount of fuelwood extracted from forests have declined, while the collection of other forest resources remained largely unchanged. While residents engaged less in cattle grazing after GTGP implementation, goat grazing and pig forage collection remained unchanged. Paired camera trap and household survey data analyses suggest that presence of humans and dogs was positively related to resource collection, and negatively related to wildlife occupancy, which likely contributed to the lower wildlife species richness detected at GTGP forest relative to natural forest. The results suggest that while PES programs, like GTGP, may reduce some human impacts on forest ecosystems (i.e. reduced firewood collection and cattle grazing) and provide habitat for wildlife through afforestation, the persistence of other types of resource use may limit the positive benefits of PES to wildlife diversity. Our results suggest there may be opportunities for PES programs to extend the multi-benefit approach of balancing human needs and ecosystem services to increase potential benefits to wildlife and biodiversity.

1. Introduction

Ecosystem services, defined as positive benefits (direct or indirect) that organisms or natural systems provide to people, including provisioning (e.g., food, water, fiber), regulating (e.g., climate, floods, disease), cultural (e.g., recreational, aesthetic, and spiritual benefits), and supporting (e.g., soil formation, photosynthesis, nutrient cycling) services, are essential to human survival and well-being (Harrison et al., 2010; Millennium Ecosystem Assessment, 2005). Human activity, including population growth, climate change, resource overexploitation, pollution, infrastructure development, and habitat loss, degradation, and fragmentation have led to a loss of biodiversity, degrading ecosystems and their corresponding ecosystem services across many natural systems (Alkemade et al., 2009; Bradshaw and Brook, 2014; Brook et al., 2008; Cardinale et al., 2012; Newbold et al., 2016, 2015). To address threats of human activities to ecosystems while sustaining socio-economic needs of human communities, many places across the world have established programs that provide payments for ecosystem services (PES) (Biedenweg and Gross-Camp, 2018). PES programs aim to protect ecosystem services while supporting sustainable livelihoods by providing financial or in-kind incentives directly to resource users to undertake environmentally desirable actions or avoid environmental damage (Jack et al., 2008; Wunder, 2013, 2007). PES conceptually captures the reciprocal relationships between human systems and ecological function and processes (Huber-Stearns et al., 2017; Lewison et al., 2017). These programs have devoted substantial efforts to PES conceptual principles, design, participation and compliance, and socio-environmental impacts (Wunder et al., 2018). Yet, considerable uncertainty still

---

[*] Corresponding author.
E-mail address: hsiangling@dragon.nchu.edu.tw (H.L. Chen).

http://dx.doi.org/10.1016/j.ancene.2020.100255

exists regarding how PES programs influence and are influenced by resource and land use (Addison and Greiner, 2016; Goh and Yanosky, 2016; Grima et al., 2018; Kronenberg and Hubacek, 2013; Lewison et al., 2017). PES programs have been linked to changes in land use and economic activities, in particular shifts in labor patterns (Li et al., 2018; Wunder, 2008). These programs have also been found to lead to indirect changes in the environment and ecosystem services (Ouyang et al., 2016). Limited research exists, however, on how changes in economic activities after the implementation of PES can affect the biodiversity of wildlife and other biodiversity related ecosystem services, such as creating wildlife habitat (Ali et al., 2018; Brouwer et al., 2011; Ingram et al., 2014; Miteva et al., 2012; Prager et al., 2016).

This study examines the Grain-to-Green Program (GTGP, also known as the Sloping Land Conversion Program and Returning Farmland to Forest Program) in China, one of the largest PES programs in the world (He and Lang, 2015; Hua et al., 2016; Liu et al., 2008), to understand how PES programs may directly affect economic activities and indirectly impact the diversity of wildlife. In China, rapid human population growth and land transformation, often combined with the overexploitation of resources, have degraded ecosystems and threatened the persistence of hundreds of species (He, 2014; Liu, 2003; Pan et al., 2017; World Bank, 2001). To address degrading environmental conditions and improve ecosystem services, the country has implemented large-scale conservation programs like GTGP. GTGP converts cropland on steep slopes to forest or grassland through afforestation or grassland restoration, while compensating participating farmers with cash and grain (Pan et al., 2017). Under the GTGP and other PES programs, economic activities such as logging, farming, mining, and infrastructure construction are also restricted. Although GTGP was developed primarily to reduce soil erosion and runoff by increasing vegetation cover, supporting species biodiversity was identified as a secondary benefit (Xu et al., 2006). GTGP has been found to increase forest cover and reduced water surface runoff and soil erosion (Liu et al., 2008; Ouyang et al., 2016). Even controlling for potentially confounding factors (e.g., income, age, gender, household assets), studies have also linked GTGP to

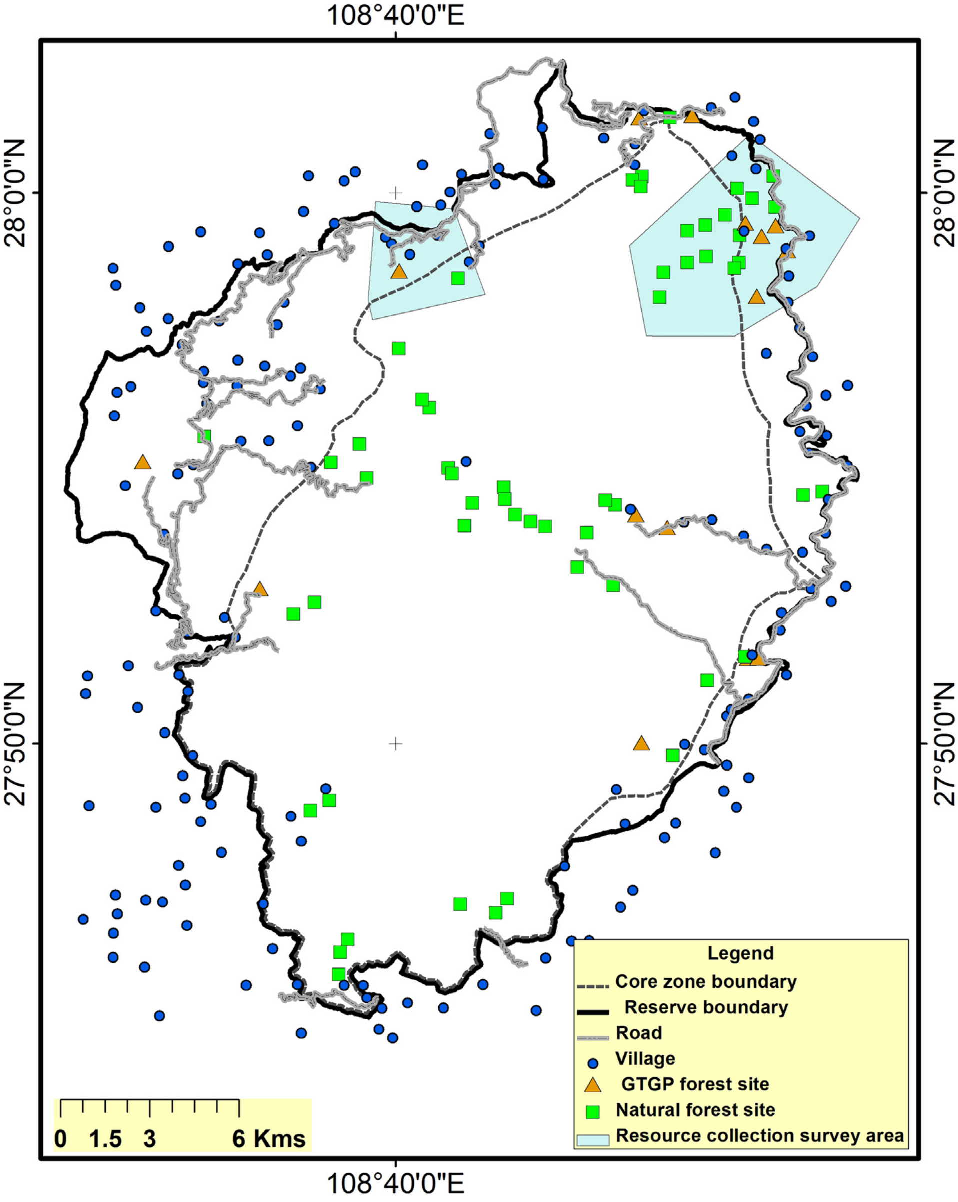

Fig. 1. Sampling plots for resource extraction and wildlife survey in areas enrolled in Grain-to Green Programs ($n = 16$) and in natural forest ($n = 55$) in Fanjingshan National Nature Reserve, China. (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article).

substantial changes in local people's livelihoods and economic activities, such as released labor from agricultural work, increased out-migration, reduced extraction of local resources, and more engagement in local off-farm jobs or businesses (Uchida et al., 2009; Zhang et al., 2018a; Zhao, 1999). In Guizhou Province alone, the number of migrant workers increased from 2.2 to 3.1 million between 2000 (before the GTGP) and 2005 (after the GTGP), representing a 48 % increase (Liu et al., 2008).

In contrast to the observed changes in forest cover, water runoff, soil erosion and agricultural labor, relatively limited research has examined how human decisions and activities translate into changes in wildlife species richness or activity (Ingram et al., 2014; Lewison et al., 2017). Thus, the degree to which GTGP has benefited wildlife diversity, as a function of changes in economic activities after PES implementation, is poorly understood. In this study, we use household surveys to document changes in human socioeconomic activities after implementation of GTGP implementation. We investigate whether reported changes in resource use and collection are associated with changes in wildlife species richness based on data from wildlife camera traps and multi-species occupancy model. We integrate social and ecological data to assess the effects of PES program on wildlife conservation. We address the following research questions:

(1) How do human socioeconomic activities, such as logging and other extractive forms of resource use change after GTGP implementation?
(2) How do these socioeconomic activities affect wildlife species richness and occupancy?
(3) How do socioeconomic activities, wildlife occupancy and richness compare in PES and natural forest sites?
(4) How may GTGP influence wildlife diversity as a function of changes in forest resource use?

We hypothesize that:

(1) GTGP implementation was associated with a decrease in extractive resource use.
(2) Extractive resource use has negative effects on wildlife occupancy and richness.
(3) Resource use is higher and wildlife occupancy and richness are lower in PES site than natural forest sites.

2. Materials and methods

2.1. Study area

Fanjingshan National Nature Reserve (FNNR) (Fig. 1), Guizhou Province, China was established in 1978 as a protected area for the endangered Grey snub-nosed monkey (*Rhinopithecus brelichi*). It was then extended to conserve other animal and plant species protected by Law of the People's Republic of China on the Protection of Wildlife, such as the Asiatic black bear (*Ursus thibetanus*), the Elliot's pheasant (*Syrmaticus ellioti*) and the dove-tree (*Davidia involucrata*) (Wu et al., 2004). The reserve spans 419 km^2 and has nearly 2000 m of vertical relief with over 95 % of forest cover, which forms the altitudinal zonation ranging from evergreen broadleaf forest to deciduous forest. The reserve contains a large amount of undisturbed primary forest compared to other areas at the same latitude and is within one of the 25 global biodiversity hotspots (Myers et al., 2000).

In 1986, FNNR became a member of UNESCO's Man and the Biosphere Programme that strives to improve the relationships between people and their environment. Over 13,000 people in 25 villages live in the reserve, of which a large proportion (70 %) are ethnic minorities like Tujia and Miao. Farming, grazing, and resource gathering take place within or near the reserve boundaries (Wandersee, 2013). Tourism and economic development, such as construction for infrastructure, are increasing in FNNR (Aitken and An, 2012). Traditionally, residents have grown crops and vegetables, raised pigs in captivity, and grazed livestock including goats and cattle in a free ranging style. For captive livestock like pigs, feed comprises crops, vegetables, and plants collected from local forests. FNNR permits local people to enter non-core areas (Fig. 1) for resource collection and fuelwood extraction. Fuelwood extraction had led to a gradual reduction of woodland in FNNR (Xiang et al., 2009).

In response to environmental degradation in the area, GTGP was initiated in FNNR in 2001. Although slope steepness of

Table 1
Response of household surveys (n = 494) about time spending on farming, collecting natural resources, raising livestock and other economic activities after implementation of Grain to Green Program in in Fanjingshan National Nature Reserve, China, 2014-2015.

Activity	Decrease (%)	Same (%)	Increase (%)	N. of household
Farming- vegetables	46.77	46.46	6.77	325
Farming - rice	67.31	26.92	5.77	156
Farming - potato	65.08	29.49	5.43	295
Farming - corn	55.69	29.27	15.04	246
Farming - sweet potato	60.65	33.21	6.14	277
Harvest bamboo	15.46	64.95	19.59	97
Harvest timber	13.68	64.21	13.68	95
Collect fuelwood	49.38	45.94	4.69	320
Collect herb	7.02	75.44	17.54	57
Collect mushroom	25.00	71.15	3.85	104
Collect bamboo shoots	15.67	72.39	11.94	134
Collect other edible plants	16.87	68.67	14.46	166
Raise pig	9.50	84.30	6.20	242
Raise cattle	33.33	56.86	9.80	102
Raise goat	4.55	84.09	11.36	44
Raise chicken and duck	6.82	84.85	8.33	132
Operate restaurant	0.00	73.07	26.93	52
Operate hotel	0.00	80.85	19.15	47
Operate transportation	0.00	76.47	23.53	51
Operate shop	4.54	88.64	6.82	44
Local labor	6.50	43.90	49.60	123
Outside labor	5.88	43.14	50.98	102

farmlands is the main criterion for inclusion in GTGP in China (Liu et al., 2008), all farmlands in FNNR regardless of steepness can be eligible for GTGP enrollment due to FNNR's standing as a national nature reserve. Each participating household receives 3583 yuan (or 512 US$ at 1 US$ = 6.99 yuan exchange rate as of July 13, 2020) per ha of converted cropland per year. The croplands enrolled in GTGP can be planted with ecological tree species (i.e., tree species primarily providing ecological functions and services), economic species such as tea trees, fruit trees, or can leave existing vegetation to grow without planting. Currently, about 55 % of households in FNNR participate in GTGP, and enrolled farmlands are mainly at elevation <2000 m, in the experimental zone (areas outside the core zone boundary, Fig. 1) of FNNR.

2.2. Human socioeconomic activity and resource use after GTGP implementation

To measure the potential effect of GTGP on resource use, we asked local residents how key economic activities in FNNR, including farming, forest resource extraction, and livestock use (Xiang et al., 2009), changed since GTGP implementation. While discerning causation versus correlation between numerical changes in human resource use activities and GTGP implementation is challenging, the linkage between GTGP and resource uses are based firmly in the robust theory of the multiphasic response (Bilsborrow, 1987; Boserup, 1965). This theory posits that rural households, when confronted by increasing population pressures, resource use constraints, or changes in institutions, may adopt livelihood or demographic responses, like deferring marriage, reducing marital fertility, out-migration, expanding the land area, intensifying agricultural production, or switching to off-farm work. Previous research at other sites enrolled in GTGP has found that local villagers tend to migrate out of the area and/or find local off-farm jobs, which has led to a reduction in local resources extracted (Uchida et al., 2009; Zhang et al., 2018a; Zhao, 1999).

In 2015, we surveyed 494 households based on a stratified random sampling of 3256 households in FNNR. We asked respondents to characterize the time spent on an array of economic activities as less, the same, or more after GTGP implementation. Economic activities included farming, forest resource collection (e.g., fuelwood, timber, herbs, mushrooms, bamboo, bamboo shoots, and other edible plants), livestock use, running business, labor in FNNR, and labor out of FNNR (Table 1). To account for inherent limitations of self-reported data in social surveys, such as inaccuracy in memory and respondents' tendency to satisfy the interviewer (Axinn et al., 1999; Bilsborrow et al., 1984), we refrained from asking respondents to report absolute amounts of resources extracted or to provide a specific data of the activity.

2.3. Human socioeconomic activity and wildlife diversity

2.3.1. Camera trapping

To investigate potential linkages of wildlife and economic activity in the nature reserve, we established 71 sampling plots across FNNR, with 55 plots in natural forest and 16 plots in GTGP forest (Fig. 1). Each plot was 20 m × 20 m. Limited accessibility due to the steep terrain and access restriction set by FNNR constrained the locations of sampling plots. Plots were established based on forest types, elevation, distance to other plots, as well as expert knowledge from FNNR staff and local field guides. The goal was to distribute the plots across the observed variation of vegetation categories in the entire FNNR. The number of plots in GTGP forest is less than plots in natural forest, because the GTGP forests are mainly located in the experimental zone of FNNR, which accounts for one third of land area of FNNR. Sampling plots were at least 1 km apart to minimize spatial autocorrelation. We recorded and classified forest types at 71 plots based on established forest categories of the FNNR: evergreen broadleaf forest ($n = 15$), mixed evergreen and deciduous forest ($n = 30$), deciduous forest ($n = 9$), bamboo ($n = 6$), and afforested conifer ($n = 11$). We deployed a Bushnell Trophy Cam infrared camera at each plot to monitor presence of humans, livestock, mammals (>0.5 kg) and pheasants (0.2–2 kg) from April 2015 to August 2016. We mounted cameras on trees from 0.3 to 1 m above the ground. Cameras that are motion-sensitive at auto-sensitivity recorded three photos upon detection, with a 1-sec delay between photographs. Checking cameras every four months assured proper functioning, change of batteries and memory cards, and retrieval of images.

2.3.2. Wildlife occupancy modeling

Multi-species hierarchical occupancy modelling (Dorazio and Royle, 2005) with a Bayesian approach (Rich et al., 2016) estimated the probability a species occurred within the area sampled by a camera station during our survey period. It also accounted for incomplete detection (MacKenzie et al., 2002). Site occupancy (also referred to as animal occurrence)—defined as the probability that a target species occupies a patch —has been widely used to address basic ecological questions related to geographic distribution, habitat relationships, resource selection and species interactions (O'Connell et al., 2011). Multi-species hierarchical occupancy models (Dorazio and Royle, 2005) integrate data across species, permitting estimates of community-level and group-level species richness and species-specific occupancy (Russell et al., 2009; Zipkin et al., 2010, 2009). Specifying models of site occupancy and detectability enables estimation of the probability of occupancy at individual camera stations. We can infer effects of environmental characteristics and human activities on animal occurrence after accounting for variation in detectability.

To avoid a high number of non-detection records (Mackenzie and Royle, 2005; Tobler et al., 2015), we treated each two-week period as a repeat survey at a particular plot, resulting in an average of 17 (SD 8.7) surveys per plot. We applied a generalized linear mixed modelling approach to incorporate site-level characteristics affecting species-specific occurrence and detection probabilities (Dorazio and Royle, 2005; Russell et al., 2009). We fitted a single model with a limited number of covariates hypothesized to influence the distributions of wildlife and evaluated how these variables affect wildlife communities (Rich et al., 2016; Zipkin et al., 2010, 2009). We interpreted probability of occurrence of a species at a camera site as probability of using the habitat at the plot during the sampling period rather than consider the site to be occupied permanently (MacKenzie et al., 2006).

To test the hypothesis that the occurrence of wildlife may be influenced by economic activity indicated by presence of humans and dogs, and presence of livestock, we quantified the presence of humans and dogs as detection rate of humans plus detection rate of dogs. Presence of livestock was quantified as detection rate of cattle plus detection rate of goats. We focused on cattle and goats because they are the most common free ranging livestock in FNNR. Detection rate for each species at each plot was calculated by dividing number of surveys with humans, dogs, cattle, and goats by total number of surveys at the plot. We also accounted for underlying differences in the forest type among plots (evergreen broadleaf forest [reference level], mixed evergreen and deciduous forest, deciduous forest, bamboo, and afforested conifer) to consider forest type as a potential variable that could explain differences in wildlife occupancy. We assumed the detection probability may be affected by distance to roads, villages and farms. To reduce the dimensionality of these distance metrics, we conducted a principal component analysis to create a combined distance measure to represent the distance to human disturbance.

This disturbance accounted for 73 % of the variation and had loadings evenly distributed among distance to roads (factor loading: 0.55), distance to villages (0.57), and distance to farms (factor loading: 0.62).

The occurrence probability for species was specified as:

logit (probability of occurrence) = $\alpha0$ + $\alpha1$(bamboo) + $\alpha2$(conifer) + $\alpha3$(mixed evergreen and deciduous) + $\alpha4$(deciduous) + $\alpha5$ (presence of humans and dogs) + $\alpha6$(presence of livestock),

and detection probability as:

logit (detection probability) = $\beta0$ + $\beta1$(*distance to human disturbance*).

Standardizing all continuous covariates generated a mean of zero and standard deviation of one. Therefore, the inverse logit of $\alpha0$ and $\beta0$ are the occupancy and detection probabilities, respectively, at a camera station in evergreen broadleaf forest and with average covariate values. The remaining coefficients of continuous covariates (i.e. $\alpha5$, $\alpha6$, $\beta1$) represent the effect of an increase of one standard deviation in the covariate value.

We linked species-specific occupancy models to a community-level occupancy model by treating species as random effects derived from a normally distributed, community-level hyper-parameter (Rich et al., 2016; Zipkin et al., 2010). For the community model, the α coefficients were modelled as $\alpha\sim$ normal (μ_α, $\sigma^2{}_\alpha$) where μ_α is the community-level mean and $\sigma^2{}_\alpha$ is the variance (Chandler et al., 2013). Because wildlife species may react to the environment differently as a function of animal type and body size, we divided wildlife into four groups based on animal type and mean body mass for males and females (Smith and Xie, 2008). The four groups were pheasants (0.5–2 kg), small (<10 kg), medium (10–50 kg) and large (>50 kg) mammals (Appendix 1). To assess group-level effects, we modeled species-specific coefficients as functions of the community-level mean, group-level mean, and species-specific random effect for group-level models (Rich et al., 2016). We estimated posterior distributions of parameters using Markov chain Monte Carlo (MCMC) implemented in JAGS (version 3.4.0) through R2Jags (Plummer, 2011) in program R. We generated three chains of 50,000 iterations after a burn-in of 10,000 and thinned by 50. For priors, we used a uniform distribution of 0–1 on the real scale for $\alpha0$ and $\beta0$ and uniform from 0 to 1 0 for σ parameters. A normal prior distribution with a mean of 0 and standard deviation of 100 on the logit-scale estimated the remaining covariate effects (Rich et al., 2016). We assessed convergence using the Gelman–Rubin statistic where values <1.1 indicated convergence (Gelman et al., 2003). We used occupancy matrices generated by MCMC iterations to estimate overall species richness and richness by wildlife groups at each camera station (Rich et al., 2016; Zipkin et al., 2010). Appendix 2 presents the detailed specification for the group model and how we calculated species richness.

2.3.3. Mapping the distribution of resource use

We used 19 of the 71 plots (6 GTGP plots, 13 natural plots) in northern FNNR (Fig. 1) to map resource use. We conducted the household survey of resource use (n = 94) at three villages near the 19 plots in 2016. Despite potential limitations of self-reported resource use data, participatory mapping is particularly important and reliable in capturing local environmental knowledge and issues related to access to land and natural resources (Aitken and Craine, 2011). We followed participatory mapping protocols developed and tested in our 2014 and 2015 surveys (An et al., 2017; Yost et al., 2020). We mapped the locations previously identified where the households collected natural resources including fuelwood, timber, herbs, mushrooms, bamboo, bamboo shoots, and other edible plants. We use ArcMap 10.1 to create 500-m buffer surrounding each plot, and counted the number of identified resource collection locations within the 500-m buffer, and used this index to represent the magnitude of resource use at each plot. We used Spearman rank-order correlation to test the strength of the relationship between detection rate of humans and dogs and the intensity of resource use at the 19 plots. The independent plots represented the larger reserve area.

2.4. Comparing resource use and wildlife in GTGP and natural forest

A t-test with unequal variance enabled comparison of the presence of humans and dogs (detection rate of humans plus detection rate of dogs), and presence of livestock (detection rate of cattle plus that of goats) between GTGP forest and natural forest. For the 19 plots with resource collection data, we used t-test to compare degree of resource extraction between GTGP and natural plots. We also used t-test to compare mean estimated species richness of wildlife between GTGP plots and natural plots. Although statistical significance in such t-tests alone does not establish any causality of GTGP (i.e., GTGP leads to higher human presence or lower wildlife species richness in some forest areas), such information may suggest potential causal links, which can be supported by theory (e.g., the theory of multiphasic response), empirical data and additional evidence from the PES literature.

3. Results

3.1. Changes in socioeconomic activity after GTGP implementation

Socioeconomic activity changed in FNNR after GTGP implementation. The reported time engaged in farming activity decreased for 59 % of households, whereas the time of operating business and labor increased (Table 1). Nearly 70 % of households reported that most of their natural resource collection did not change after GTGP implementation, except for fuelwood collection, which 50 % of households reported spending less time (Table 1). The majority (~85 %) of households spent the same amount of time raising pigs, goats, chickens and ducks, but one third of households reported to have engaged less in cattle grazing (Table 1).

3.2. Resource use and wildlife diversity

We detected 19 species of wildlife over 16,579 trap nights at 68 of 71 plots, with the remaining three sites having zero detections due to equipment loss or damage. Detected species included four species of pheasants, six species of small, six species of medium, and three species of large mammals (Appendix 1). Because distinguishing between Indian muntjac (*Muntiacus muntjak*) and Reeves's muntjac (*Muntiacus reevesi*) was difficult, we grouped these two species as “muntjac”. Among 19 species, 10 species are either protected under Law of the People’s Republic of China on the Protection of Wildlife, or listed as Endangered, Vulnerable or Nearly Threatened on the IUCN Red List of Threatened Species (Version 2016-3). Camera station-specific estimates of species richness ranged from 3 to 13 species (Appendix 3), with a mean of eight species. Among forest types, mean species richness was lower at bamboo (5 species SE [0.75]) and coniferous forest (5 species SE [0.55]), which were the dominant forest types at GTGP forest (80 %, n = 16), and was highest in deciduous forest (10 species SE [0.65], Fig. 2a), which was missing in GTGP forest.

The mean probability of occupancy across all species and camera stations was 0.30 (SD 0.07, 95 % CI: 0.17–0.45), ranging from 0.07 for Grey snub-nosed monkey and muntjac to 0.79 for wild boar (*Sus scrofa*). The mean detection probability across all species was 0.11 (SD 0.02, 95 % CI: 0.07–0.16), ranging from 0.01 for Asian black bear to 0.37 for muntjac (Appendix 1). At community

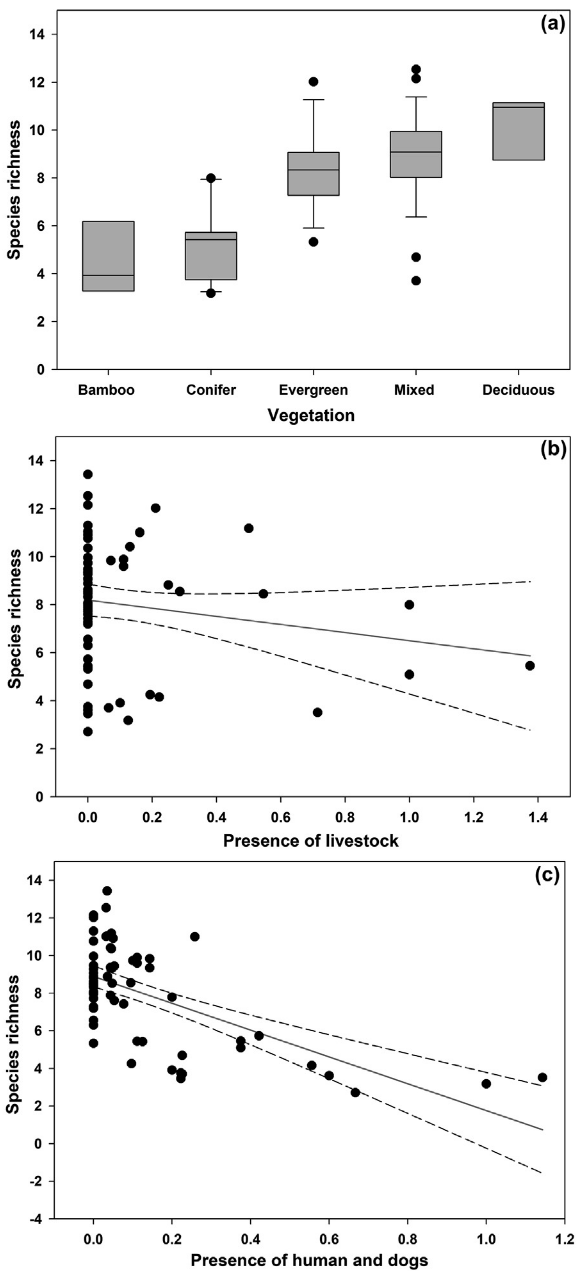

Fig. 2. Mean estimated wildlife species richness among forest types: bamboo, afforested conifer, evergreen broadleaf forest, mixed evergreen and deciduous forest, and deciduous forest (a), in relation to detection rate of livestock (b), detection rate of humans and dogs (c) in Fanjingshan National Nature Reserve, China, 2015-2016.

level, probability of occupancy was lower at bamboo and coniferous forest comparing to at evergreen broadleaf forest, the reference level (Table 2). Nine of the 19 species had higher probability of occupancy in mixed evergreen and deciduous forest, and fourteen species had greater probability of occupancy in deciduous forest, comparing to that in evergreen broadleaf forest (Appendix 1).

We found differences in the effect of resource use and other socioeconomic activities on wildlife occupancy and richness. Although occupancy did not vary significantly with presence of cattle and goats on community level (Fig. 2b), occupancy of Tibetan macaque (*Macaca thibetana*) and the tufted deer (*Elaphodus cephalophus*) was negatively related to livestock presence (i.e. 95 % CI did not overlap zero) (Appendix 1). In contrast, presence of humans and dogs was negatively correlated with occupancy across the wildlife community (Fig. 2c), especially for Golden pheasant (*Chrysolophus pictus*) and Temminck's tragopan (*Tragopan temminckii*). For the 19 subsampled plots with resource collection data, detection rate of humans and dogs was positively related to the intensity of resource collection ($r = 0.52$, $p = 0.024$, Fig. 3a). We also found that the intensity of resource collection was negatively correlated with wildlife species richness ($r =-0.67$, $p < 0.001$, Fig. 3b). As expected, the precision of estimates was lower for species with limited numbers of detections, leading to diffuse posterior distributions for their estimates of covariate effects. The Gelman–Rubin statistics indicated convergence for all parameters (all value <1.1, with an average of 1.005).

3.3. Comparing resource use and wildlife diversity in GTGP and natural forest

As expected, humans and dogs were more likely to be present at GTGP plots than at natural forest plots ($t_{66} = 3.07$, $p = 0.007$, Fig. 4a). Detection of cattle and goats was not significantly different between GTGP plots and natural plots ($t_{66} = 1.36$, $p = 0.19$, Fig. 4a). We recorded greater intensity of resource collection (i.e. number of resource collection locations within 500 m of each camera) at GTGP plots than natural forest ($t_{17} = 3.83$, $p = 0.001$, Fig. 4b). Wildlife species richness was lower at GTGP forest across all four species groups ($t_{66} = 4.97$, $p < 0.001$, Fig. 4c). Mean species richness was 6 species (SE 0.55) at GTGP forest plots and was 9 species (SE 0.30) at natural forest plots. This relationship was consistent even when we only included plots in the experimental zone (11 natural forest plots and 12 GTGP forest plots, $t_{21} = 2.76$, $p = 0.01$), where most GTGP forest in FNNR located.

4. Discussions

Although biodiversity related ecosystem services are not typically the primary drivers of PES implementation, many PES

Table 2
Mean and 95 % confidence interval estimates of the community-level and group-level hyper-parameters hypothesized to influence the probability of occurrence (α) and detection (β) of 19 species of pheasants and mammals in Fanjingshan National Nature Reserve, China 2015-2016. Species groups included pheasants, small (<10 kg), medium (10–50 kg) and large (>50 kg) mammals. Bold text indicates the 95 % confidence interval does not overlap with zero.

Variables	Community	Pheasant	Small mammal	Medium mammal	Large mammal
Bamboo (α1)	−0.73 (−1.04, 0.20)	−0.73 (−3.47, 1.39)	−1.00 (−3.34, 1.03)	−1.11 (−3.31, 0.72)	1.73 (−3.83, 12.71)
Conifer (α2)	−0.60 (−1.76, 0.48)	−0.15 (−3.22, 2.72)	−0.55 (−3.49, 2.48)	−1.32 (−4.53, 1.07)	0.66 (−4.58, 8.32)
Mixed evergreen /deciduous (α3)	0.43 (−0.09, 0.95)	**1.15 (0.32, 2.02)**	−0.72 (−1.64, 0.12)	**1.07 (0.31, 1.87)**	−0.38 (−2.04, 1.35)
Deciduous (α4)	0.99 (−0.09, 2.15)	**3.89 (1.62, 7.44)**	−0.11 (−2.25, 1.95)	0.77 (−1.11, 2.66)	4.03 (−2.38, 19.90)
Presence of human and dogs (α5)	−**0.62 (−1.16, -0.12)**	−1.11 (−2.29, 0.12)	−0.63 (−1.83, 0.50)	−0.56 (−1.57, 0.36)	−1.44 (−4.78, 1.71)
Presence of livestock (α6)	−0.19 (−0.84, 0.42)	0.31 (−1.33, 1.97)	−0.25 (−1.71, 1.18)	−0.46 (−1.80, 0.87)	0.53 (−2.34, 3.98)
Distance to human disturbance (β1)	0.09 (−0.07, 0.27)	0.11 (−0.19, 0.44)	0.23 (−0.12, 0.58)	−0.04 (−0.31, 0.29)	0.09 (−0.42, 0.53)

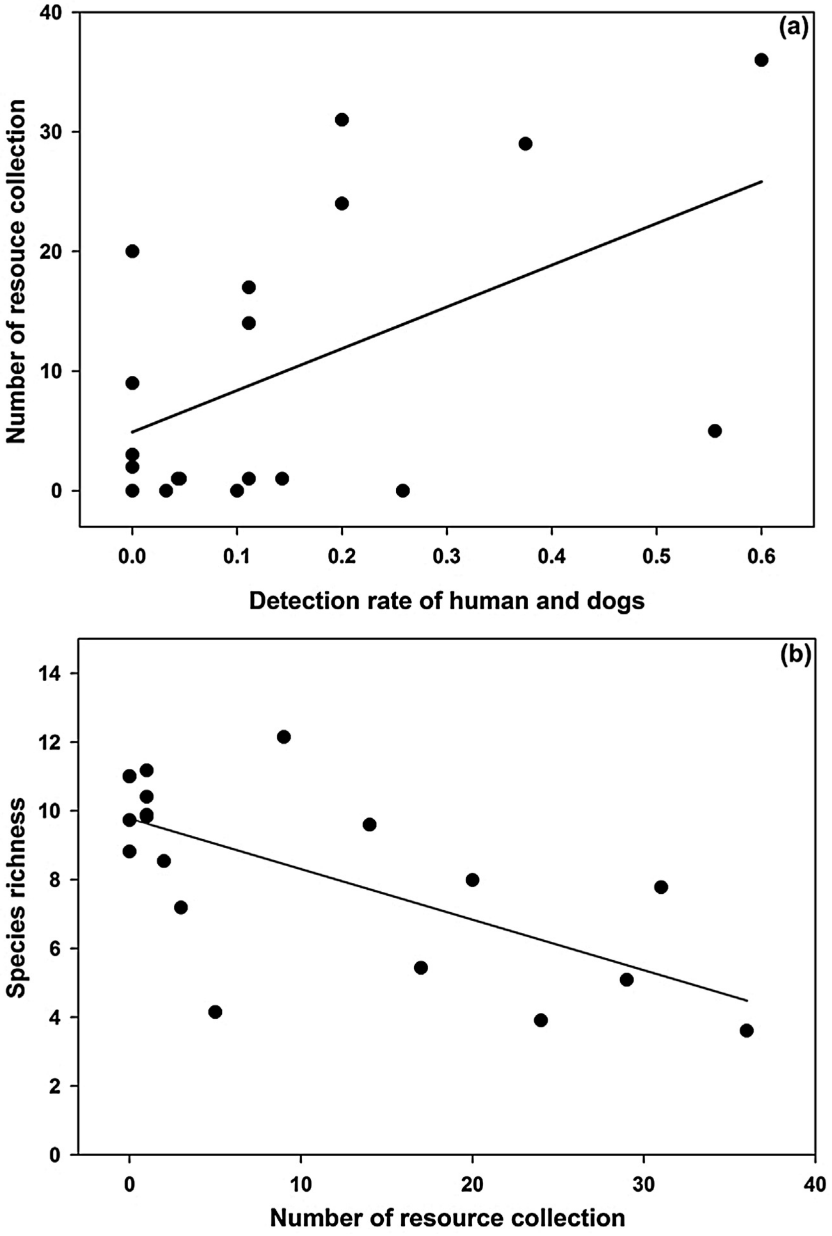

Fig. 3. Relationship between detection rate of humans and dogs and the intensity of resource collection (a), and relationship between mean estimated wildlife species richness and the intensity of resource collection (b) in Fanjingshan National Nature Reserve, China, 2015-2016.

programs, including GTGP, have identified supporting species biodiversity as a secondary goal (Xu et al., 2006; Lewison et al., 2017). Results of this study therefore help to address existing knowledge gaps into the direct and indirect effects of GTGP, one of the largest PES programs, on wildlife biodiversity-related ecosystem services.

Without counterfactual study sites or a controlled before-after-control-impact (BACI) design, directly measuring how GTGP, specifically reported changes in resource use and collection since GTGP implementation, has impacted wildlife is challenging. In the study site, as in most PES locations worldwide, data does not exist to allow for a controlled or counterfactual comparison between conditions before and after relative to PES policies. In light of this constraint, our aim was to use all existing data, in this case household survey and wildlife presence and occupancy data, to consider how GTGP may directly and indirectly effect wildlife diversity in FNNR. Empirical and household data suggest that GTGP has influenced resource source in FNNR, has likely had direct positive effects on wildlife through afforestation related habitat improvements, as well as indirect negative effects through continued resource use.

4.1. Resource use and socioeconomic activity after GTGP implementation

The results were consistent with changes in local human communities observed in other nature reserves in China, such as Wolong Nature Reserve (Chen et al., 2014; Yang et al., 2018). From 2000 to 2014, the number of new migrants (i.e., migrants who first migrated out in a certain year) increased consistently every year but one, and was positively correlated to the accumulated amount of afforested land (Tsai et al., 2016; An et al., 2017). Research on PES has commonly reported that GTGP implementation have reduced demand for agriculture labor, and increased local laborers'

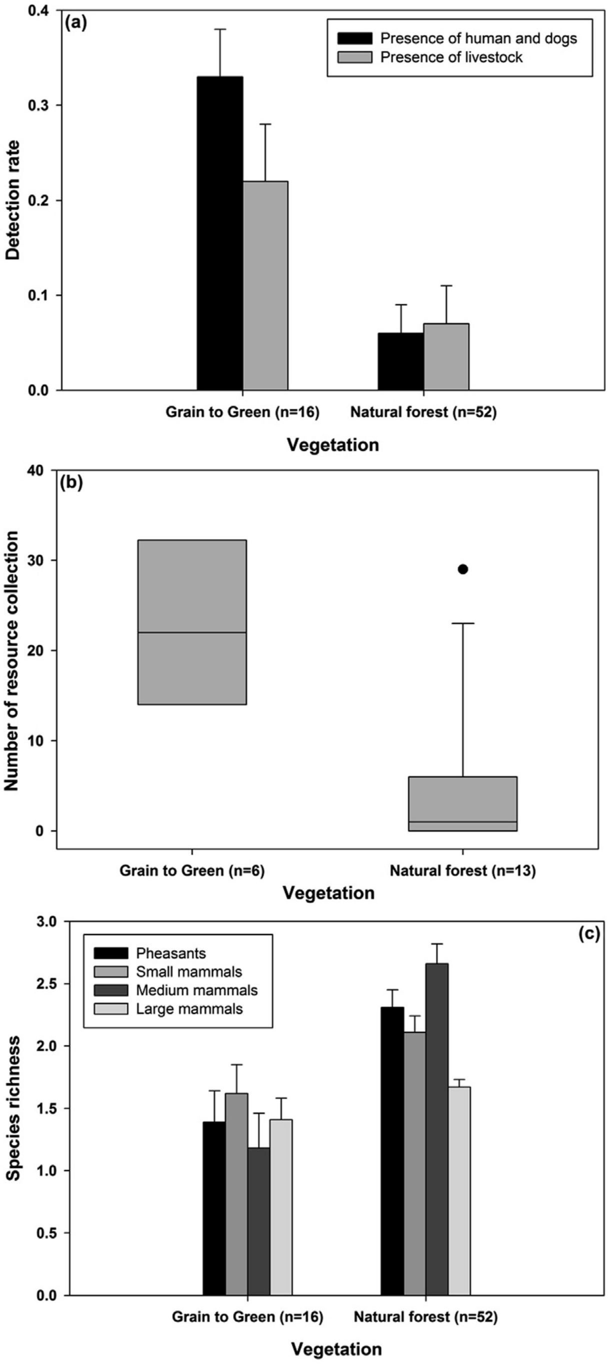

Fig. 4. Comparisons between Grain-to Green forests and natural forests: detection rate of livestock and detection rate of humans and dogs (a), intensity of resource collection (b), mean estimated wildlife species richness (c). (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article).

likelihood of outmigration to cities (e.g. Dai et al., 2009; Dao et al., 2018).

The survey results suggested that the reported amount of forest fuelwood extracted declined after GTGP initiation, a finding also consistent with research from other areas in China where GTGP has been implemented (Chen et al., 2014; Tuanmu et al., 2016). While this decline may be influenced by a number of possible pathways (e.g., outmigration of young people from the reserve, Yost et al., 2020), that GTGP implementation contributed to the observed reduction in fuelwood consumptions is likely. Although electricity is a preferred alternative when family income permits (Xiang et al., 2009), most FNNR households currently use electricity mainly for lighting and some electronic appliances, but not for heating and cooking (Wandersee et al., 2012). In FNNR, households with higher fuelwood consumption tended to have lower awareness of human impacts on environment (Wandersee et al., 2012), so the reduction on fuelwood collection could reflect an increased awareness of human impacts on forest resources, affordability of electricity or outmigration related to GTGP program participation.

Despite reported changes in economic and resource use activities after GTGP initiation, results from FNNR also suggested that some types of resources use did not change after GTGP implementation. These activities include the collection of timber for construction, bamboo harvesting, and collection of edible plants such as bamboo shoots, mushrooms, herbs, and other plants. Furthermore, the data suggest that the intensity of resource collection was higher in GTGP forest (Fig. 4b), likely because timber and bamboo harvest are allowable at GTGP forest with permission, and limited extraction of other forest resources, excluding endangered or protected species, is not tightly regulated.

Similarly, these findings suggested that, although local people have changed resource use patterns with the implementation of GTGP, other resource uses that impact the forest, which include non-cattle livestock grazing and collection of livestock forage remained the same. In FNNR, most households own a small group (<10) of livestock to provide meat and to generate a variable amount of income. Goats, like cattle, are primarily free ranging and graze on forest year-round and local households spend two to three hours per day on average to collect plants in forest for pigs. Although compared to farming, livestock grazing and rearing is on a relatively small scale in FNNR, the dependence of livestock on forest resources should not be overlooked.

The increased income and labor shifts might lead to positive and negative effects on ecosystem services and more broadly for wildlife conservation(Zhang et al., 2018a, 2018b). GTGP participation was a positive predictor of outmigration at FNNR (An et al., 2020; Yost et al., 2020), which likely reduces the amount of some resource use activities and the corresponding pressure on local forest ecosystem. The GTGP-induced raise in income and the time released from farming could, however, also lead to an increase in forest resource collection, increased economic and tourism development in FNNR (Aitken and An, 2012; Wandersee et al., 2012), higher demand for handcrafts, edible plants, and timber for building larger houses and infrastructure. All of these outcomes can negatively impact the sustainability of forest resources.

4.2. The direct and indirect effects of resource use on wildlife diversity

The results suggested that GTGP has directly increased wildlife habitat through afforestation of farmlands, i.e., forest cover in FNNR has increased since GTGP implementation (Tsai et al., 2016). It has also indirectly increased wildlife habitat by changing patterns of local villagers' forest resource use, like fuelwood extraction and reductions in cattle grazing. The results also suggest that reduced cattle grazing after GTGP implementation was associated with a positive effect on wildlife occupancy, especially for Tibetan macaque and the tufted deer (classified as Nearly Threatened by IUCN).

Although evidence indicated that GTGP reduced some activities related to resource use, household surveys also suggested that other economic activities were persistent after implementation of GTGP, particularly forest resource extraction and livestock grazing. The results suggested that the degree of human disturbance at GTGP forest was higher than in natural forest, which is likely, in part, a function of proximity (i.e., GTGP parcels are often close to human residences). While GTGP parcels reduce direct disturbance from humans in natural forests by attracting a large amount of human subsistence related activities, this access to GTGP sites may limit the positive impact of PES on wildlife diversity because of the persistence of human and livestock activities.

These analyses suggested that wildlife species richness at GTGP forests was lower than that in natural forests, likely due to both human activities and differences in forest type between GTGP and natural forests. Forest type, in terms of their ecological and economic value for GTGP, has been identified as an important variable that can directly influence the ecological outcomes from PES programs (He and Lang, 2015). Despite the option to grow ecologically important tree species on farmlands enrolled in GTGP, most GTGP sites are planted with economically valuable tree species such as Chinese red pine (*Pinus massoniana*), Chinese fir (*Cunninghamia lanceolata*), and bamboo (*Phyllostachys heterocycla cv pubescens*) (Chen et al., 2020). Therefore, 80 % of GTGP plots were bamboo or coniferous forest, which tend to have lower wildlife species richness (Fig. 3b).

Study results indicated that forest type alone does not explain the lower wildlife species richness in GTGP sites. Resource collection was positive associated with presence of humans and dogs, and the probability of animal occurrence and wildlife species richness were lower in the presence of humans and dogs, even after accounting for effect of forest type (Fig. 4c). The finding that cattle and goats, related to the lower occupancy of Tibetan macaque and the tufted deer, concurs a study in South America which found species richness of amphibians was low in areas of cattle grazing after implementation of a PES program (Basham et al., 2016). Given that cattle and goats were present at one third of sampling plots, free ranging livestock may have a broad impact on the wildlife community in FNNR and point to an important consideration for PES programs.

5. Conclusion

In conclusion, study of the Grain-to-Green Program in Fanjingshan National Nature Reserve in China produced answers to the research questions, as follows:

(1) GTGP implementation was accompanied by a reported reduction in farming, the amount of fuelwood extraction and cattle grazing in forests, while other livestock use and collection of timber wood, bamboo, and edible plants did not decline after GTGP implementation.
(2) Presence of human and dogs that likely associated with resource use had negative effects on wildlife occupancy and richness. The intensity of resource collection was negatively correlated with wildlife species richness. Presence of livestock had negative effects on occupancy of specific species, like the Tibetan macaque and the tufted deer.
(3) Humans and dogs were more likely to be present at GTGP forest than at natural forest, as well as forest resource collection. The lower wildlife species richness observed at GTGP forest compared to natural forests was likely influenced by the level of human disturbance.
(4) GTGP may directly increase wildlife habitat through afforestation of farmlands and indirectly by changing patterns of local villagers' forest resource use. Reduced farming, extraction of fuelwood, and cattle grazing at PES forest may increase wildlife diversity. However, the persistence of other economic activities at PES forest like collection and use of forest resource, and livestock raising including pigs and goats may limit the positive impact of GTGP on wildlife diversity.

As local and regional governments in China have made concerted efforts to balance protecting ecosystem services with resource use and development, PES programs and concurrent shifts to tourism and other economic activities such as dairy, cattle and deer farming have offset displaced rural agriculture and timber industries (Liu et al., 2008; Xu et al., 2006). The study provides additional evidence that PES programs, like GTGP, affect biodiversity-related ecosystem services directly and indirectly, including the habitats and species richness of wildlife. Robust empirical evidence exists that FNNR has experienced an increase in forest cover (Tsai et al., 2016) and concurrent reported reductions in some types of resource use since GTGP implementation. This may have resulted from the widely implemented PES program, GTGP, but could also have been driven by concurrent changes in livelihood strategies unrelated to GTGP.

The need for the multi-benefit approach of PES programs continues to intensify as human populations and resource limitation grow (Engel et al., 2008). For PES programs that also support biodiversity conservation, a need further exists to improve design, development and implementation of PES programs to strike a better balance among human well-being, conservation, and economic development (Engel et al., 2008). This balance includes accounting for complex human preferences, responses and social impacts of PES programs (Bowles and Polanía-Reyes, 2012; Rode et al., 2015). This study underscores the importance of a greater understanding of the underlying drivers and pathways that govern the complex linkages among cultural, socioeconomic, demographic, and ecological factors of GTGP and PES programs worldwide, in recognition of the full range of ecosystem services needed to maintain viability human and natural systems.

Declaration of Competing Interest

The authors declare no competing interests.

Acknowledgements

Household surveys were conducted under permits from the San Diego State University's Institutional Review Board (Protocol # 1732093). Field efforts were conducted under permits from the San Diego State University's Institutional Animal Care and Use Committee (Protocol # 14-01-002 L). We acknowledge funding from the National Science Foundation under the Dynamics of Coupled Natural and Human Systems program (DEB-1212183), San Diego State University, and the equipment support by IDEA WILD. This project would not have been possible without the logistic support by W. Xu of the Chinese Academy of Sciences, and The Administration Office of Fanjingshan National Nature Reserve, especially D. F. Shia, D. C. Shia, and W. Dong. We thank stuff, students, and volunteers of San Diego State University and Tongren College for processing data and assisting field work. We also thank the anonymous reviewers and the handling editor for their constructive comments and suggestions.

Appendix A. Supplementary data

Supplementary material related to this article can be found, in the online version, at doi:https://doi.org/10.1016/j.ancene.2020.100255.

References

Addison, J., Greiner, R., 2016. Applying the social–ecological systems framework to the evaluation and design of payment for ecosystem service schemes in the Eurasian steppe. Biodivers. Conserv. 25, 2421–2440. doi:http://dx.doi.org/10.1007/s10531-015-1016-3.

Aitken, S.C., An, L., 2012. Figured worlds: environmental complexity and affective ecologies in Fanjingshan, China. Ecol. Modell. 229, 5–15. doi:http://dx.doi.org/10.1016/j.ecolmodel.2011.05.024.

Aitken, S., Craine, J., 2011. Affective geovisualizations. In: Dodge, M., Kitchin, R., Perkins, C. (Eds.), The Map Reader. John Wiley & Sons, Inc, Hoboken, New Jersey, USA.

Ali, M., Kennedy, C.M., Kiesecker, J., Geng, Y., 2018. Integrating biodiversity offsets within circular economy policy in China. J. Clean. Prod. 185, 32–43. doi:http://dx.doi.org/10.1016/j.jclepro.2018.03.027.

Alkemade, R., vanOorschot, M., Miles, L., Nellemann, C., Bakkenes, M., tenBrink, B., 2009. GLOBIO3: a framework to investigate options for reducing global terrestrial biodiversity loss. Ecosystems 12, 374–390. doi:http://dx.doi.org/10.1007/s10021-009-9229-5.

An, L., Lewison, R., Bilsborrow, R., Aitken, S., Stow, D., 2017. Final report of NSF CNH Project "Impacts of Ecosystem Service Payments in Coupled Natural and Human Systems" (2012-2016). San Diego, CA, USA.

An, L., Mak, J., Yang, S., Lewison, R., Stow, D.A., Chen, H.L., Xu, W., Shi, L., Tsai, Y.H., 2020. Cascading impacts of payments for ecosystem services in complex human-environment systems. JASSS 23, 5. doi:http://dx.doi.org/10.18564/jasss.4196.

Axinn, W.G., Pearce, L.D., Ghimire, D., 1999. Innovations in life history calendar applications. Soc. Sci. Res. 28, 243–264. doi:http://dx.doi.org/10.1006/ssre.1998.0641.

Basham, E.W., Gonzalez del Pliego, P., Acosta-Galvis, A.R., Woodcock, P., Medina Uribe, C.A., Haugaasen, T., Gilroy, J.J., Edwards, D.P., 2016. Quantifying carbon and amphibian co-benefits from secondary forest regeneration in the Tropical Andes. Anim. Conserv. 19, 548–560. doi:http://dx.doi.org/10.1111/acv.12276.

Biedenweg, K., Gross-Camp, N.D., 2018. A brave new world: integrating well-being and conservation. Ecol. Soc. 23, art32. doi:http://dx.doi.org/10.5751/ES-09977-230232.

Bilsborrow, R.E., 1987. Population pressures and agricultural development in developing countries: a conceptual framework and recent evidence. World Dev. 15, 183–203. doi:http://dx.doi.org/10.1016/0305-750X(87)90077-5.

Bilsborrow, R.E., Oberai, A.S., Standing, G., 1984. . Migration Surveys in Low-income Countries: Guidelines for Survey and Questionnaire Design. Routledge Kegan & Paul, London, UK.

Boserup, E., 1965. The Conditions of Agricultural Growth: the Economics of Agrarian Change under Population Pressure. Aldine Publishing Company, Chicago.

Bowles, S., Polanía-Reyes, S., 2012. Economic incentives and social preferences: substitutes or complements? J. Econ. Lit. 50, 368–425. doi:http://dx.doi.org/10.1257/jel.50.2.368.

Bradshaw, C.J.A., Brook, B.W., 2014. Human population reduction is not a quick fix for environmental problems. Proc. Natl. Acad. Sci. U. S. A. 111, 16610–16615. doi:http://dx.doi.org/10.1073/pnas.1410465111.

Brook, B.W., Sodhi, N.S., Bradshaw, C.J.A., 2008. Synergies among extinction drivers under global change. Trends Ecol. Evol. (Amst.) 23, 453–460. doi:http://dx.doi.org/10.1016/j.tree.2008.03.011.

Brouwer, R., Tesfaye, A., Pauw, P., 2011. Meta-analysis of institutional-economic factors explaining the environmental performance of payments for watershed services. Environ. Conserv. 38, 380–392. doi:http://dx.doi.org/10.1017/S0376892911000543.

Cardinale, B.J., Duffy, J.E., Gonzalez, A., Hooper, D.U., Perrings, C., Venail, P., Narwani, A., Mace, G.M., Tilman, D., A.Wardle, D., Kinzig, A.P., Daily, G.C., Loreau, M., Grace, J.B., Larigauderie, A., Srivastava, D.S., Naeem, S., 2012. Biodiversity loss and its impact on humanity. Nature 489, 326. doi:http://dx.doi.org/10.1038/nature11373.

Chandler, R.B., King, D.I., Raudales, R., Trubey, R., Chandler, C., Arce Chávez, V.J., 2013. A small-scale land-sparing approach to conserving biological diversity in tropical agricultural landscapes. Conserv. Biol. 27, 785–795. doi:http://dx.doi.org/10.1111/cobi.12046.

Chen, X., Viña, A., Shortridge, A., An, L., Liu, J., 2014. Assessing the effectiveness of payments for ecosystem services: an agent-based modeling approach. Ecol. Soc. 19, art7. doi:http://dx.doi.org/10.5751/ES-05578-190107.

Chen, H.L., Lewison, R.L., An, L., Tsai, Y.H., Stow, D., Shi, L., Yang, S., 2020. Assessing the effects of payments for ecosystem services programs on forest structure and species biodiversity. Biodivers. Conserv. 1–18. doi:http://dx.doi.org/10.1007/s10531-020-01953-3.

Dai, L., Zhao, F., Shao, G., Zhou, L., Tang, L., 2009. China's classification-based forest management: procedures, problems, and prospects. Environ. Manage. 43, 1162–1173. doi:http://dx.doi.org/10.1007/s00267-008-9229-9.

Dao, T.H., Docquier, F., Parsons, C., Peri, G., 2018. Migration and development: dissecting the anatomy of the mobility transition. J. Dev. Econ. 132, 88–101. doi:http://dx.doi.org/10.1016/J.JDEVECO.2017.12.003.

Dorazio, R.M., Royle, J.A., 2005. Estimating size and composition of biological communities by modeling the occurrence of species. J. Am. Stat. Assoc. 100, 389–398. doi:http://dx.doi.org/10.1198/016214505000000015.

Engel, S., Pagiola, S., Wunder, S., 2008. Designing payments for environmental services in theory and practice: an overview of the issues. Ecol. Econ. 65, 663–674. doi:http://dx.doi.org/10.1016/j.ecolecon.2008.03.011.

Gelman, A., Carlin, J.B., Stern, H.S., Rubin, D.B., 2003. Bayesian Data Analysis. Chapman and Hall. Chapman and Hall, Boca Raton, FL doi:http://dx.doi.org/10.1017/CBO9781107415324.004.

Goh, T.Y., Yanosky, A., 2016. Payment for Ecosystem Services works, but not exactly in the way it was designed. Glob. Ecol. Conserv. 5, 71–87. doi:http://dx.doi.org/10.1016/j.gecco.2015.11.005.

Grima, N., Singh, S.J., Smetschka, B., 2018. Improving payments for ecosystem services (PES) outcomes through the use of Multi-Criteria Evaluation (MCE) and the software OPTamos. Ecosyst. Serv. 29, 47–55. doi:http://dx.doi.org/10.1016/j.ecoser.2017.11.019.

Harrison, P.A., Vandewalle, M., Sykes, M.T., Berry, P.M., Bugter, R., deBello, F., et al., 2010. Identifying and prioritising services in European terrestrial and freshwater ecosystems. Biodivers. Conserv. 19, 2791–2821. doi:http://dx.doi.org/10.1007/s10531-010-9789-x.

He, J., 2014. Governing forest restoration: local case studies of sloping land conversion program in Southwest China. For. Policy Econ. 46, 30–38. doi:http://dx.doi.org/10.1016/j.forpol.2014.05.004.

He, J., Lang, R., 2015. Limits of state-led programs of payment for ecosystem services: field evidence from the Sloping Land Conversion Program in southwest China. Hum. Ecol. 43, 749–758. doi:http://dx.doi.org/10.1007/s10745-015-9782-9.

Hua, F., Wang, X., Zheng, X., Fisher, B., Wang, L., Zhu, J., Tang, Y., Yu, D.W., Wilcove, D. S., 2016. Opportunities for biodiversity gains under the world's largest reforestation programme. Nat. Commun. 7, 12717. doi:http://dx.doi.org/10.1038/ncomms12717.

Huber-Stearns, H.R., Bennett, D.E., Posner, S., Richards, R.C., Fair, J.H., Cousins, S.J.M., Romulo, C.L., 2017. Social-ecological enabling conditions for payments for ecosystem services. Ecol. Soc. 22, art18. doi:http://dx.doi.org/10.5751/ES-08979-220118.

Ingram, J.C., Wilkie, D., Clements, T., McNab, R.B., Nelson, F., Baur, E.H., Sachedina, H. T., Peterson, D.D., Foley, C.A.H., 2014. Evidence of Payments for Ecosystem Services as a mechanism for supporting biodiversity conservation and rural livelihoods. Ecosyst. Serv. 7, 10–21. doi:http://dx.doi.org/10.1016/j.ecoser.2013.12.003.

Jack, B.K., Kousky, C., Sims, K.R.E., 2008. Designing payments for ecosystem services: lessons from previous experience with incentive-based mechanisms. Proc. Natl. Acad. Sci. U. S. A. 105, 9465–9470. doi:http://dx.doi.org/10.1073/pnas.0705503104.

Kronenberg, J., Hubacek, K., 2013. Could payments for "ecosystem services create an ecosystem service curse"? Ecol. Soc. 18, art10. doi:http://dx.doi.org/10.5751/ES-05240-180110.

Lewison, R.L., An, L., Chen, X., 2017. Reframing the payments for ecosystem services framework in a coupled human and natural systems context: strengthening the integration between ecological and human dimensions. Ecosyst. Heal. Sustain. 3, 5.

Li, H., Cai, Y., Zhang, Y., Liu, Y., Zhang, K., Yang, X., 2018. Impact of a cross-jurisdictional Payment for Ecosystem Services program on the participants' welfare in North China. J. Clean. Prod. 189, 454–463. doi:http://dx.doi.org/10.1016/j.jclepro.2018.04.081.

Liu, J., 2003. Protecting China's biodiversity. Science 300, 1240–1241.

Liu, J., Li, S., Ouyang, Z., Tam, C., Chen, X., 2008. Ecological and socioeconomic effects of China's policies for ecosystem services. Proc. Natl. Acad. Sci. U. S. A. 105, 9477–9482. doi:http://dx.doi.org/10.1073/pnas.0706436105.

Mackenzie, D.I., Royle, J.A., 2005. Designing occupancy studies: general advice and allocating survey effort. J. Appl. Ecol. 42, 1105–1114. doi:http://dx.doi.org/10.1111/j.1365-2664.2005.01098.x.

MacKenzie, D.I., Nichols, J.D., Lachman, G.B., Droege, S., Royle, J.A., Langtimm, C.A., 2002. Estimating site occupancy rates when detection probabilities are less than one. Ecology 83, 2248–2255. doi:http://dx.doi.org/10.2307/3072056.

MacKenzie, D.I., Nichols, J.D., Royle, J.A., Pollock, K.H., Bailey, L.L., Hines, J.E., 2006. Occupancy Estimation and Modeling: Inferring Patterns and Dynamics of Species Occurrence. Elsevier Inc. Elsevier Academic Press, London doi:http://dx.doi.org/10.1017/CBO9781107415324.004.

Millennium Ecosystem Assessment, 2005. Ecosystems and Human Well-being: Biodiversity Synthesis. World Resources Institute, Washington, D.C., USA.

Miteva, D.A., Pattanayak, S.K., Ferraro, P.J., 2012. Evaluation of biodiversity policy instruments: what works and what doesn't? Oxford Rev. Econ. Policy 28, 69–92. doi:http://dx.doi.org/10.1093/oxrep/grs009.

Myers, N., Mittermeier, R.A., Mittermeier, C.G., daFonseca, G.A.B., Kent, J., 2000. Biodiversity hotspots for conservation priorities. Nature 403, 853–858. doi:http://dx.doi.org/10.1038/35002501.

Newbold, T., Hudson, L.N., Hill, S.L.L., Contu, S., Lysenko, I., Senior, R.A., Börger, L., Bennett, D.J., Choimes, A., Collen, B., Day, J., DePalma, A., Díaz, S., Echeverria-Londoño, S., Edgar, M.J., Feldman, A., Garon, M., Harrison, M.L.K., Alhusseini, T., Ingram, D.J., Itescu, Y., Kattge, J., Kemp, V., Kirkpatrick, L., Kleyer, M., Correia, D.L. P., Martin, C.D., Meiri, S., Novosolov, M., Pan, Y., Phillips, H.R.P., Purves, D.W., Robinson, A., Simpson, J., Tuck, S.L., Weiher, E., White, H.J., Ewers, R.M., Mace, G. M., Scharlemann, J.P.W., Purvis, A., 2015. Global effects of land use on local terrestrial biodiversity. Nature 520, 45–50. doi:http://dx.doi.org/10.1038/nature14324.

Newbold, T., Hudson, L.N., Arnell, A.P., Contu, S., DePalma, A., Ferrier, S., Hill, S.L.L., Hoskins, A.J., Lysenko, I., Phillips, H.R.P., Burton, V.J., Chng, C.W.T., Emerson, S., Gao, D., Pask-Hale, G., Hutton, J., Jung, M., Sanchez-Ortiz, K., Simmons, B.I., Whitmee, S., Zhang, H., Scharlemann, J.P.W., Purvis, A., 2016. Has land use pushed terrestrial biodiversity beyond the planetary boundary? A global assessment. Science 353, 288–291. doi:http://dx.doi.org/10.1126/science.aaf2201.

O'Connell, A.F., Nichols, J.D., Karanth, K.U., 2011. Camera Traps in Animal Ecology. Springer, Japan doi:http://dx.doi.org/10.1007/978-4-431-99495-4.

Ouyang, Z., Zheng, H., Xiao, Yi, Polasky, S., Liu, J., Xu, W., Wang, Q., Zhang, L., Xiao, Yang, Rao, E., Jiang, L., Lu, F., Wang, X., Yang, G., Gong, S., Wu, B., Zeng, Y., Yang, W., Daily, G.C., 2016. Improvements in ecosystem services from investments in natural capital. Science 352, 1455–1459. doi:http://dx.doi.org/10.1126/science.aaf2295.

Pan, X., Xu, L., Yang, Z., Yu, B., 2017. Payments for ecosystem services in China: policy, practice, and progress. J. Clean. Prod. 158, 200–208. doi:http://dx.doi.org/10.1016/j.jclepro.2017.04.127.

Plummer, M., 2011. JAGS: A Program for the Statistical Analysis of Bayesian Hierarchical Models by Markov Chain Monte Carlo. .

Prager, C.M., Varga, A., Olmsted, P., Ingram, J.C., Cattau, M., Freund, C., Wynn-Grant, R., Naeem, S., 2016. An assessment of adherence to basic ecological principles by payments for ecosystem service projects. Conserv. Biol. 30, 836–845. doi:http://dx.doi.org/10.1111/cobi.12648.

Rich, L.N., Miller, D.A.W., Robinson, H.S., McNutt, J.W., Kelly, M.J., Hayward, M., 2016. Using camera trapping and hierarchical occupancy modelling to evaluate the spatial ecology of an African mammal community. J. Appl. Ecol. 53, 1225–1235. doi:http://dx.doi.org/10.1111/1365-2664.12650.

Rode, J., Gómez-Baggethun, E., Krause, T., 2015. Motivation crowding by economic incentives in conservation policy: a review of the empirical evidence. Ecol. Econ. 117, 270–282. doi:http://dx.doi.org/10.1016/j.ecolecon.2014.11.019.

Russell, R.E., Royle, J.A., Saab, V.A., Lehmkuhl, J.F., Block, W.M., Sauer, J.R., 2009. Modeling the effects of environmental disturbance on wildlife communities: avian responses to prescribed fire. Ecol. Appl. 19, 1253–1263. doi:http://dx.doi.org/10.1890/08-0910.1.

Smith, A.T., Xie, Y., 2008. A Guide to the Mammals of China. Princeton University Press, Princeton, New Jersey, USA.

Tobler, M.W., Zúñiga Hartley, A., Carrillo-Percastegui, S.E., Powell, G.V.N., 2015. Spatiotemporal hierarchical modelling of species richness and occupancy using camera trap data. J. Appl. Ecol. 52, 413–421. doi:http://dx.doi.org/10.1111/1365-2664.12399.

Tsai, Y.H., Stow, D., Shi, L., Lewison, R., An, L., 2016. Quantifying canopy fractional cover and change in Fanjingshan National Nature Reserve, China using multi-temporal Landsat imagery. Remote Sens. Lett. 7, 671–680. doi:http://dx.doi.org/10.1080/2150704X.2016.1177243.

Tuanmu, M.N., Viña, A., Yang, W., Chen, X., Shortridge, A.M., Liu, J., 2016. Effects of payments for ecosystem services on wildlife habitat recovery. Conserv. Biol. 30, 827–835. doi:http://dx.doi.org/10.1111/cobi.12669.

Uchida, E., Rozelle, S., Xu, J., 2009. Conservation payments, liquidity constraints, and off-farm labor: impact of the Grain-for-Green Program on rural households in China. Am. J. Agric. Econ. 91, 70–86. doi:http://dx.doi.org/10.1111/j.1467-8276.2008.01184.x.

Wandersee, S.M., 2013. Land-cover and Land-use Change in Human-environment Systems: Understanding Complex Interactions among Policy and Management, Livelihoods, and Conservation. University of California, Santa Barbara.

Wandersee, S.M., An, L., López-Carr, D., Yang, Y., 2012. Perception and decisions in modeling coupled human and natural systems: a case study from Fanjingshan National Nature Reserve. China. Ecol. Modell. 229, 37–49. doi:http://dx.doi.org/10.1016/j.ecolmodel.2011.08.004.

World Bank, 2001. The World Bank China: Air, Land, and Water. World Bank, Washington DC.

Wu, G., Wang, H., Fu, H., Zhao, J., Yang, Y., 2004. Habitat selection of Guizhou golden monkey (*Phinopithecus roxellanae brelichi*) in Fanjing Mountain Biosphere Reserve, China. J. For. Res. 15, 197–202.

Wunder, S., 2007. The efficiency of payments for environmental services in tropical conservation. Conserv. Biol. 21, 48–58. doi:http://dx.doi.org/10.1111/j.1523-1739.2006.00559.x.

Wunder, S., 2008. Payments for environmental services and the poor: concepts and preliminary evidence. Environ. Dev. Econ. 13, 279–297. doi:http://dx.doi.org/10.1017/S1355770X08004282.

Wunder, S., 2013. When payments for environmental services will work for conservation. Conserv. Lett. 6, 230–237. doi:http://dx.doi.org/10.1111/conl.12034.

Wunder, S., Brouwer, R., Engel, S., Ezzine-De-Blas, D., Muradian, R., Pascual, U., Pinto, R., 2018. From principles to practice in paying for nature's services. Nat. Sustain. 1, 145–150. doi:http://dx.doi.org/10.1038/s41893-018-0036-x.

Xiang, Z.F., Nie, S.G., Lei, X.P., Chang, Z.F., Wei, F.W., Li, M., 2009. Current status and conservation of the gray snub-nosed monkey *Rhinopithecus brelichi* (Colobinae) in Guizhou, China. Biol. Conserv. 142, 469–476. doi:http://dx.doi.org/10.1016/j.biocon.2008.11.019.

Xu, J., Yin, R., Li, Z., Liu, C., 2006. China's ecological rehabilitation: unprecedented efforts, dramatic impacts, and requisite policies. Ecol. Econ. 57, 595–607. doi:http://dx.doi.org/10.1016/j.ecolecon.2005.05.008.

Yang, H., Lupi, F., Zhang, J., Chen, X., Liu, J., 2018. Feedback of telecoupling: the case of a payments for ecosystem services program. Ecol. Soc. 23, art45. doi:http://dx.doi.org/10.5751/ES-10140-230245.

Yost, A., An, L., Bilsborrow, R., Shi, L., Chen, X., Yang, S., Zhang, W., 2020. Mechanisms behind concurrent payments for ecosystem services in a Chinese nature reserve. Ecol. Econ. 169, 106509 doi:http://dx.doi.org/10.1016/j.ecolecon.2019.106509.

Zhang, Q., Bilsborrow, R.E., Song, C., Tao, S., Huang, Q., 2018a. Determinants of out-migration in rural China: effects of payments for ecosystem services. Popul. Environ. 40, 182–203. doi:http://dx.doi.org/10.1007/s11111-018-0307-5.

Zhang, Q., Song, C., Chen, X., 2018b. Effects of China's payment for ecosystem services programs on cropland abandonment: A case study in Tiantangzhai Township, Anhui, China. Land Use Policy 73, 239–248. doi:http://dx.doi.org/10.1016/j.landusepol.2018.01.001.

Zhao, Y., 1999. Leaving the countryside: rural-to-Urban migration decisions in China. Am. Econ. Rev. 89, 281–286. doi:http://dx.doi.org/10.2307/117121.

Zipkin, E.F., DeWan, A., Andrew Royle, J., 2009. Impacts of forest fragmentation on species richness: a hierarchical approach to community modelling. J. Appl. Ecol. 46, 815–822. doi:http://dx.doi.org/10.1111/j.1365-2664.2009.01664.x.

Zipkin, E.F., Andrew Royle, J., Dawson, D.K., Bates, S., 2010. Multi-species occurrence models to evaluate the effects of conservation and management actions. Biol. Conserv. 143, 479–484. doi:http://dx.doi.org/10.1016/j.biocon.2009.11.016.