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Disaggregating livelihood dependence on ecosystem services to inform land management

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ABSTRACT

The direct links between ecosystem services and households' well-being have long been discussed, but documentation remains sparse. This paper outlines a practical method for estimating the value of ecosystem services as they contribute to household livelihoods in rural regions. Measuring these links at the household level enables disaggregated assessment of how ecosystem services relate to household wellbeing. To demonstrate the method, we use a unique dataset of 1749 households in northern China to show how livelihoods depend on ecosystem services, focusing on the ecosystem contributions to goods they sell or consume. We disaggregate the household-level value of ecosystem services across locations and to various beneficiary groups, which shows substantial variation in dependency on different types of services. These results have practical implications for land management strategies by safeguarding the most critical ecosystems and targeting management goals for beneficiaries. Organizations can use these methods to better inform policy design and understand who will win and lose from proposed programs.

1. Introduction

The promise and potential of the ecosystem services (ES) paradigm is that it provides a theoretical and conceptual framing directly linking the natural environment and human well-being (Daily, 1997). Ecosystem services are the myriad ways that nature benefits people, ranging from how the environment contributes to the provisioning of livelihoods, to the regulation and support of environmental systems as a foundation for basic life, to ways in which we culturally value nature that help fulfill human life (Millennium Ecosystem Assessment, 2005). Recognizing and quantifying these links has helped remove their "economic invisibility" in policy and decision-making (Sukhdev, 2010), and an ES rationale is being used to justify policy from the national (Bouwma et al., 2018; Schaefer et al., 2015) to municipal (Hansen et al., 2015) levels.

Most studies that demonstrate ES′ role in our economy and society have almost exclusively calculated its total social value – its value aggregated to some regional level or higher (Nelson et al., 2009; Polasky et al., 2011; Troy and Wilson, 2006). These studies provide important information for understanding policy options, including evaluation of tradeoffs (Zheng et al., 2016). Yet at these aggregate scales it is difficult to design targeted payment for ecosystem services schemes (Reed et al., 2014), identify where tradeoffs occur among policy options (Li et al., 2015), evaluate equity issues at the forefront of sustainability debates and the Sustainable Development Goals (Griggs et al., 2013; Schröter et al., 2017), or address the role of access and property rights in procuring landscape benefits (Robinson et al., 2018; Wieland et al., 2016). At more resolute scales, like the household level, we could better measure the direct relationship between ES and human well-being. Such a finer-scaled perspective on how ES are used and valued is needed to improve efficiency and incorporate equity into ES policy.

Past work has predominantly focused on the biophysical supply of ES (Boerema et al., 2017; Laterra et al., 2016; Wolff et al., 2015), but understanding the demand side is critical for valuing ES in ways that can be integrated into decision-making and support vulnerable populations (Lewis and Wu, 2015; Schröter et al., 2012; Villamagna et al., 2013). Further disaggregation of the flow of ES to households and household production demonstrates how ES directly contributes to human well-being, and under what conditions (Daw et al., 2011; Ferraro et al., 2011; Howe et al., 2014; Suich et al., 2015; Tallis et al., 2008; Wieland et al., 2016). A core challenge has been to measure the contribution of multiple ES to human well-being at a small enough scale to allow for this disaggregation of benefits, over space or demographic group of interest, in a way that consistently parses out the value of ecosystems from other inputs to production (Rieb et al., 2017).

This paper presents a practical method for estimating the value of

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ecosystem services as they directly contribute to the provision of household livelihoods as one core dimension of human well-being, especially in developing regions. Using standard market and non-market valuation methods from environmental economics, we characterize the contribution (or flow) of a range of "final" ES to beneficial goods or products (Fisher et al., 2009), while separating out the value attributable to other inputs (namely, labor and manufactured materials) necessary to produce that good or service. To demonstrate the method, we estimate ES dependence among 1749 households in a critical area of water supply north of Beijing, China. We show how estimating ES dependence at the household level can reveal reliance on different types of ES by location, by type of dominant livelihood, and by demographic groups of interest. Understanding these empirical patterns has direct implications for land management.

2. Measuring ecosystem services' contribution to livelihoods

2.1. ES contributions to household production

People depend on nature's free goods and services in many ways. Ecosystems and social systems interact to co-produce ultimate benefits to human well-being (Bennett et al., 2015; Fischer and Eastwood, 2016). Fig. 1 presents a framework showing how ES can contribute to the goods by which households earn their livelihoods, or the means by which a household earns a living (Chambers and Conway, 1992; Ellis, 1999).

Starting from the right side of Fig. 1, we identify the ultimate benefits (material or non-material goods, services, or commodities (Pascual et al., 2017)) that households produce which have direct inputs from ES. We define these as goods that have not gone through other value-added steps in a production process that may have already incorporated the ES value into a market value (Sjaastad et al., 2005).

Each good is produced from various sources of inputs represented on the left side of the diagram, with flows indicated by the arrows in the middle of Fig. 1. We divide inputs into those that flow from natural capital as ecosystem services (E), and those that come from anthropogenic labor and other manufactured sources (together, non-ES inputs W). Ecosystem service inputs E represented by the green arrows are "final" services that directly contribute to producing an ultimate beneficial good or product j that is consumed or enjoyed by society (Boyd and Banzhaf, 2006; Fisher et al., 2009).

For even the simplest end benefits that we actively procure, like wild harvested non-timber forest products, we must still reconcile the value that ES contribute to the product (E_j) with the opportunity cost of labor needed to obtain it (W_j) (Fig. 1, arrow B). More capital-intensive processes and equipment are needed to acquire other wild products like fish (Fig. 1, arrow C). In other cases, like agricultural products, we must also account for the value of manufactured inputs, like fertilizer or feed, as well as ES inputs like soil fertility (Fig. 1, arrow D). Over all livelihood activities chosen by a household (gray arrows in Fig. 1, indicating household labor) that lead to benefits j, household i's total livelihood L_i can be described as a function of ecosystem (green arrows) and non-ecosystem (gray and orange arrows) inputs $L_i = f(E_{ij}, W_{ij})$.

We make the simplifying assumption that livelihoods are separable in ecosystem service and non-ecosystem service inputs such that $L_i = \sum_j (E_{ij} + W_{ij})$. We measure E_{ij} and W_{ij} in monetary terms, noting some values of E_{ij} may be unrecognized shadow values, making L_i the implicit total value of a household's livelihood.

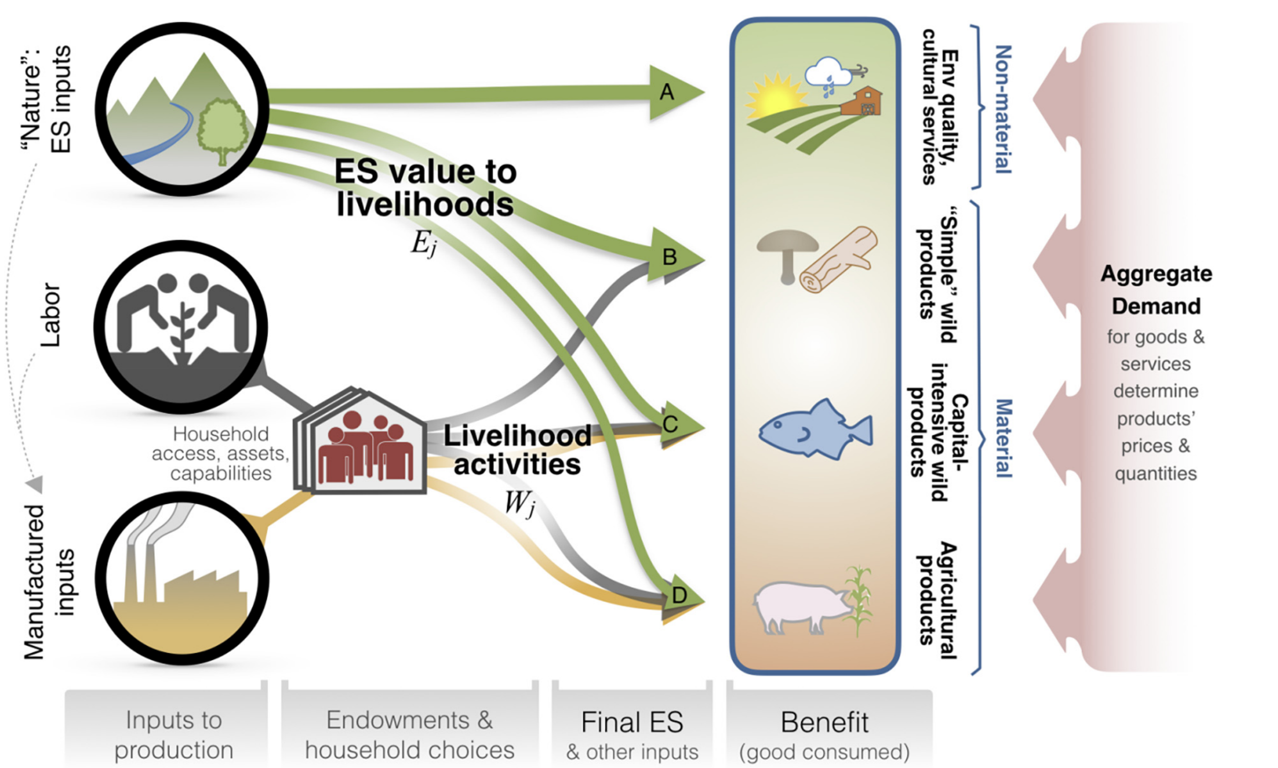

Fig. 1. The contribution of ecosystem services to livelihoods. Goods from which people benefit are made up of a variety of inputs. ES can, for example, contribute to producing goods along with human labor used to procure them and, sometimes, other manufactured inputs. Households make livelihood activity choices based on their assets, access, and capabilities, and the ecosystem endowments they get from the local landscape. Household choices determine their own demand for ecosystem services as inputs to these livelihood activities. This paper focuses on assessing the final ES, measuring the contribution (or flow) of the natural environment toward producing goods by which households "get by". Aggregate societal demand helps determine prices and therefore incentives for households to produce various quantities of the resulting goods and services consumed. (For interpretation of the references to color in this figure legend, the reader is referred to the web version of this article.)

Fig. 2. Study location and sites. The Miyun Reservoir watershed is in NW China, just north of the city of Beijing. About 1/3 of the watershed lies within the municipality of Beijing, with the rest in Hebei Province. Our 15 study villages (white dots) are distributed throughout the watershed.

2.2. Estimating ecosystem service values

We approximate ES value to a household by building on non-market valuation first principles and applying resource rent theory in the household production framework. In the simplest formulation, E_{ij} can be approximated by $E_{ij} = p_j q_{ij}$, where p_j is the implicit marginal value of ecosystem service's contribution to some ultimate benefit j, and q_{ij} is the quantity used by household i (Brown, 2017; Jackson et al., 2014; Shone and Caviglia-Harris, 2006). In the context of developing regions, we can often obtain estimates of q_{ij} through direct inspection, aggregate statistics, or household surveys. Some ES contributions are not directly traded in markets, so that the marginal value p_j is not observed. In a production context, the marginal value of an ES could be valued as an input into a household production function for each product j of interest, in which we regress the value of the product on the quantities of the inputs (Barbier, 2007; Bockstael and Freeman, 2005). While the production function approach proves a statistical way of estimating marginal values, data on quantities of ES inputs, especially at the household level in developing regions (e.g., water applied to crops, quality of soil fertility, etc.), is generally not available. This prohibits the use of a production function approach in most household settings. Alternatively, ES can often be thought of as quality components that are "weak complements" for a marketed good or service (Mäler, 1974). Yet weak complementarity in a household production framework has been used only for singular ES like recreation models (e.g., Bockstael and Freeman, 2005; Fenichel et al., 2018), forest services as weak complements to labor (e.g., Pattanayak and Butry, 2005), or at the landscape scale (e.g., Tan-Soo et al., 2016; Vincent et al., 2016). Data challenges similar to the production function approach arise in addition to having data to characterize househld livelihoods over the full suite of relevant ES.

Here we use a *resource rent* approach to estimate the value of ES. The concept of resource rent is well established in resource economics. In the environmental income literature (Sjaastad et al., 2005), natural capital accounting (Asafu-Adjaye et al., 2005; Remme et al., 2015; Sumarga et al., 2015; UN et al., 2012), and classic bioeconomics (Clark, 1990), resource rent is the value *attributable to nature* from the production of a resource. While these literatures are not always consistent with terminology of welfare economics (Jensen et al., 2015) resource rents are broadly profits that remain after accounting for other input costs from labor and other capital-derived inputs. Cavendish (2000) uses a similar calculation to estimate what he calls a "natural habitat value" for resources that are inputs to goods co-produced between nature and people.

The resource rent approach, as used here, attributes surplus profit to ES. With our simplifying assumption of separability in ecosystem service and non-ecosystem service inputs $L_i = \sum_j (E_{ij} + W_{ij})$, then if we know the value of L_i and non-ES inputs W_{ij}, we can implicitly calculate the value of ecosystem services E_{ij}. Of course there may be other reasons some households enjoy surplus profit relative to others, which include managerial skill or other unobservable characteristics that contribute to total factor productivity, which would be captured in our estimates of surplus profit. However, on average, the value of resource rent across households in a location or subgroup should give a reasonable estimate of ES value to production (Remme et al., 2015). While this is still likely a second-best approach to estimating marginal value in a household production function, when input quantities of ES are not available the resource rent approach bounds ecosystem services value to a reasonable range. In our view this is still a dramatic improvement over a current approach in much of the literature that often assigns net revenue, e.g., the total value of agricultural production, as a proxy for ES value.

3. Case study: the Miyun Reservoir watershed

3.1. Study area

We apply our methods to data from the Miyun Reservoir watershed (Fig. 2). The watershed is about 100 km north of the city of Beijing and covers 15,788 km^2, approximately the size of the State of Connecticut, with a population of near one million people. Over 90% of households are involved in agriculture. The Miyun reservoir provides Beijing with about half of its freshwater and is the only surface water source for domestic use (Kröger et al., 2012). Numerous land use policies are in effect in the watershed to protect water quality and quality, which involves tradeoffs in ES provisions for inhabitants of the watershed versus residents in Beijing (Zheng et al., 2016).

3.2. Survey

We conducted structured household surveys in 15 villages in the Miyun Reservoir watershed during the summer of 2014 and 2015. We selected villages based on three criteria: (a) the total estimated village population was relatively small such that we could reasonably expect to sample most households in the village in an attempt to characterize a village's total resource use, (b) the village resided in the headwaters of a watershed, so that their ES were not mediated by other upstream populations, and (c) the land uses in the village were predominantly rural. We used township level population estimates (National Bureau of Statistics Rural Social and Economic Investigation Corps, 2010) and areal imagery (ESRI, 2014) to guide village selection, with the aid of local officials.

The questionnaires gathered quantitative data on demographics, assets, and livelihood activities from 1749 households. In each village, we aimed to gather as close to a census of the village population as possible, with coverage rates we estimate to be around 80%. The survey elicited time budgets and income sources from each household to calculate quantitative measure of the total livelihood L_i. Time budgets detailed how households allocate labor as an input into all livelihood

Table 1
Calculation of ES values from survey data.

Ecosystem Service	Resource rent calculation
Forest resources	
Fuelwood	(kg of fuelwood collected in local forests * village average fuelwood price) − (opportunity cost of collection time)
Wild products (mushrooms, herbs, etc.)	(Household-estimated value of wild products harvested) − (opportunity cost of collection time)
Orchard	
Tree production of nuts & fruits	(Household-estimated quantity of resource harvested * sale price) − (opportunity cost of maintenance and collection time)
Agriculture	
Water, Soil	(Agricultural revenue) − (value of anthropogenic & manufactured inputs)
Pests, etc. (negative ES)	Pesticide expenditures
Livestock	
Water, Soil, Feed	(Livestock revenue) − (value of anthropogenic & manufactured inputs)
Landscape-based	
Nature-based tourism	(Wages from tourism activities) − (the opportunity cost of labor)
ES Policy	Household-reported eco-compensation payment
SLCP (tuigenhuanlin)	Reported subsidies/payments
PLDL (daogaihan)	Reported subsidies/payments
Shelterbelt program	Reported subsidies/payments
Other programs/ subsidies	Reported subsidies/payments
Non-ES dependent	
Non-ES primary industry income	Reported wages (minus any value attributed to ecosystem services)
Secondary industry	Reported wages
Tertiary industry	Reported wages
Gifts/transfers/remittances	Reported value
Rental income	Reported income
Interest income	Reported income
Welfare receipts	Reported subsidies/payments

activities, including those that relate to ES and/or employment in other (non-ES related) primary, secondary, or tertiary sectors of the economy. The survey recorded income related to these activities or whether the activity related to self-consumption, and other potential sources of income, such as gifts or remittances, that might contribution to a household's overall livelihood. All data were compiled and subsequently analyzed in Stata 15 (StataCorp, 2016).

Table 1 summarizes the livelihood categories and methods we use to account for ES in our study (the survey instrument is available upon request). The livelihood categories reflect the mix of activities in the region (Peng et al., 2017). Important non-material benefits from ES, such as cultural or psychological values, require different methods (Gould et al., 2015) that are not employed in this study, but could add another dimension with which to further understand how the benefits of ES are distributed across populations.

3.3. Case study analysis

Our analysis separates livelihood values into ecosystem and non-ecosystem derived components. Resource rent values, E_{ij}, that are calculated as "direct from ES" (see Table 1) are forest resources such as fuelwood and collection of wild products minus the labor needed to collect them; the value of orchard, agriculture, and livestock minus the cost of labor and/or manufactured inputs; the value of income from nature tourism minus the opportunity cost of labor; and the direct value of any income received from payment for ecosystem service policies. We include policy payments here because, from the household's perspective, these are a direct benefit from keeping ES intact. Values included in the "not from ES" category are income from the primary (e.g., agricultural income minus ES value), secondary, or tertiary sectors as elicited from a time budget which included 20 different occupational categories; gifts, transfers or remittances; rental or interest income from investments; or any other welfare payments or subsidies received.

We examine two metrics that describe ES value to households. First, we look at the absolute value of ES as they contribute to livelihoods E_i. We also estimate the proportion of a household's livelihood that is dependent on ES as $D_i = \frac{\sum_j E_{ij}}{L_i}$. In addition to being appropriately bound by the value of the goods households consume, our estimate of ES dependence D_i helps compare beneficiary groups that might come from different settings or have different socioeconomic statuses (Gowdy, 2004; Lele and Srinivasan, 2013).

At the village level, we aggregate households' activities and ES values to better understand how reliance on ES can differ across locations. We also examine household level ES dependence through scatterplots and distributional analysis of the relationship between ES and livelihoods for vulnerable subpopulations. Data to divide our population into groups comes directly from the household survey and includes elderly households, those made up of more than 50% members over 65; sick households, those with a chronic health condition; and a measure of socioeconomic status as defined by asset index score we calculate from our data as follows.

We construct an asset index by taking the first principle component from a principle components analysis on the number of common assets a household owns (Filmer and Pritchett, 2001). As long as the assets in our dataset are positively correlated (or that negatively correlated assets are removed), the first principle component is a vector described by various weights on the assets that have relative strength in explaining the variation over the entire catalog of assets for our population. This method reduces the dimensionality for a suite of variables that can be used to collectively summarize household socio-economic status. Asset indices have been shown to perform favorably in explaining welfare as compared to expenditure data (Sahn and Stifel, 2003), have been used widely to explore poverty dynamics (Adato et al., 2006), and are common in analyses of household activity in developing areas (Fisher et al., 2005; Robinson, 2016).

To collectively see which household factors are associated with ES dependence, we also employ simple ordinary least-squares linear regression models. These household factors come directly from our survey and include variables such as household education, household health (whether anyone in the household indicates a chronic or severe illness or disability), the ratio of those that are greater than 65 years old to the total household size, the number of children in school, the total amount of loans taken on in the past 5 years, and our asset index as an indicator of socioeconomic status.

4. Case results

4.1. Village- and household-level dependency

At an aggregate village level, Fig. 3 shows that livelihoods derived from ES vary with local-level conditions. Fig. 3a shows the range of average dependence across the villages in our sample, and Fig. 3b unpacks the contributions from various sources that comprise the "direct from ES" value following the categorization in Table 1. For example, village 6 has the largest average ES dependency at ~30%, but this is largely due to policy payments (Fig. 3b) which, in this village, are for forest protection. Village 16, in contrast, has a much different profile: less than 10% of livelihoods on average come from ES (Fig. 3a) and the sources of ES are much more diverse: ES that support agriculture and orchards, and those from forests play a more active role in people's lives (Fig. 3b).

Household-level estimates of ES dependency D_i as a percent of livelihood are plotted in Fig. 3c against the total livelihood value L_i, while Fig. 3d shows the absolute value of ES E_i against non-ES

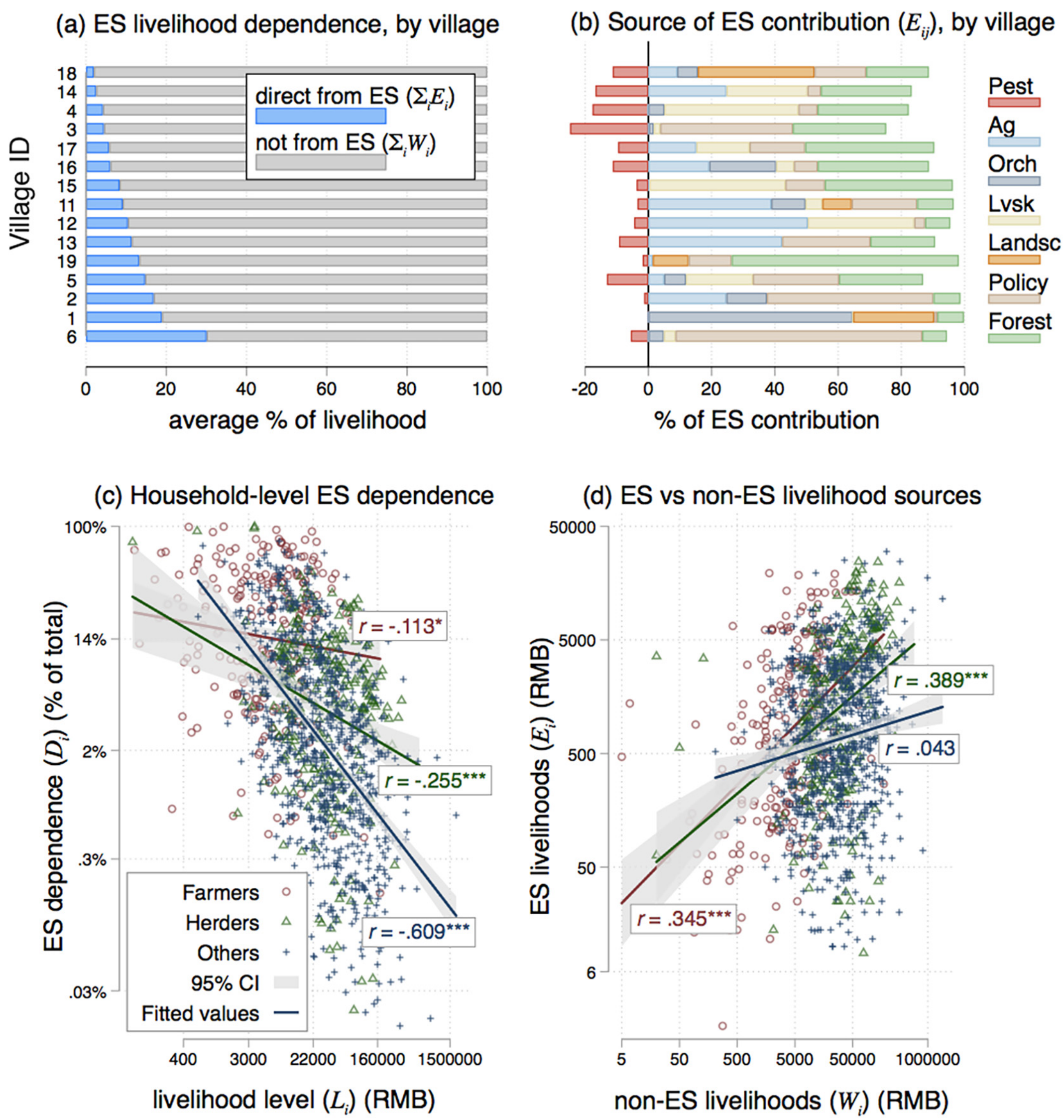

Fig. 3. Ecosystem service contributions to livelihoods. ES contributions to livelihoods vary among all the villages in our sample (a). The types of ES that matter also differ greatly by location (b) and dominant livelihood activity (c, d). At the household level, those with lower levels of livelihood support depend more on ES as a share of their livelihood (c), however, the total value of ES and non-ES contributions to livelihoods is positively correlated (d).

livelihood values W_i. Points on each plot represent a household, whose symbol denotes their dominant livelihood source as crop production ("farmers": $n = 276$), livestock production ("herders": $n = 311$), or other wage-earning activities ("others": $n = 1163$). The solid lines on the plot shows the linear predicted value of the total livelihood L_i by household types with a 95% confidence interval surrounding the predicted linear fit in shaded gray. Pearson's correlation coefficients for these fitted lines are shown color-coded by household type.

When looking at ES dependence D_i in Fig. 3c, our data show that poorer households derive a larger share of their livelihood from ES than wealthier households (as indicated by L_i). There is a strong and significant negative relationship between a household's livelihood level L_i and D_i, which is strongest for those that are not predominantly farmers or herders ($r = -0.61$, $p < 0.001$). We also see in Fig. 3c that, on average, farmers have a higher level of dependence on ES ($\mu = 22.8\%$) than herders ($\mu = 7.9\%$) or others ($\mu = 8.3\%$). The relative importance of ES appears almost reversed when looking at the value of ES that contribute to household livelihoods (E_i) relative to non-ES livelihood contributions (W_i) in Fig. 3d. We see that, especially as farmers and herders begin to utilize greater levels of non-ES inputs, their utilization of ES inputs also grows ($r = 0.35$ and 0.39, respectively, both with $p < 0.001$). Households that do not predominantly engage in farming or herding show a much weaker but still positive relationship ($r = 0.04$, $p = 0.15$) between ES and non-ES value utilization.

Some of these differences can also be seen in Fig. 4, which gives an example of how different household types relate to ES. Fig. 4a shows that different livelihood types rely on ES from various sources. Of course, it makes sense that populations that derive most of their income from farming rely more on ES values in crops and orchard production, and that herders rely on ES embedded in livestock production, but we can also see, for example, that most ES from forests go to households that are not predominantly farmers or herders. Fig. 4b organizes ES flows by wealth quintiles, as measured by our asset index. The poorest in our sample use less ES, and the proportions are different. For example, the poor rely more heavily on forest resource value as a percent of ES contribution to livelihood, but get relatively less value from ES to agriculture or orchards compared to wealthier households. Households across all wealth quintiles enjoy similar values from ES policy.

4.2. ES dependency by demographic groups

Measuring household level ES dependence also allows for

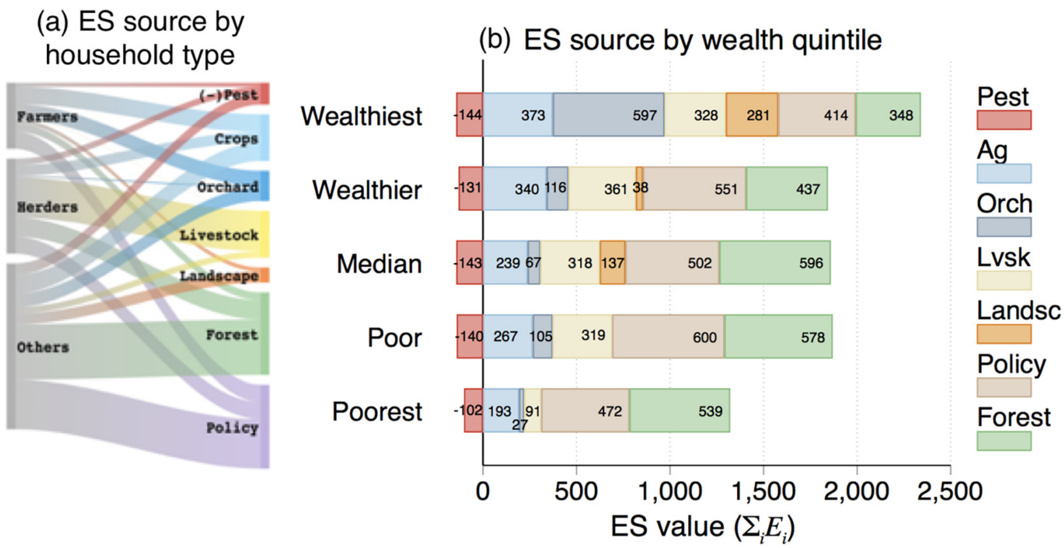

Fig. 4. Ecosystem service contributions to (a) livelihood types and (b) wealth quintiles. Ecosystem service use varies by household types. (a) Famers (n = 276), Herders (n = 311), and "Others" (n = 1163) use different proportions of ES in support of livelihoods. (b) The poorest use less and different proportions of ES.

disaggregation how ES directly affect different populations. Current environmental policy in China, and elsewhere, prioritizes protecting local landscapes to support those that are most vulnerable. Thus, understanding how various groups depend on ES allows for development of targeted land use and environmental policies to support marginalized or vulnerable populations. Distributional analyses over various demographic groups are shown in "strip plots" (Cox, 2003) in Fig. 5. These plots show a box plot and an accompanying vertical histogram of the distribution of the data for a category, giving a clear picture of the median and interquartile range of the data via the box plot, but also the spread of the data via the histogram.

Fig. 5 again compares differences in households by dependence on ES (D_i) and total ES value (E_i) in the left and right columns, respectively. We look at difference in these measures across three groups: elderly households (Fig. 5aand b), households in good health (Fig. 5c and d), and households with high or low asset indices (Fig. 5e and f). Interestingly, we see statistically significant differences (at greater than the $p < 0.001$ confidence level) in the percent dependence across all these categories (Fig. 5a, c and e), but no differences at any conventional level of statistical significance for the absolute value of ES (Fig. 5b, d and f). Thus, for example, older, sicker and poorer household *depend* more on ES for livelihoods, but these demographic groups derive about the same amount of value from ES on average.

4.3. Factors associated with dependency

The bivariate correlations described above show how different groups rely on ES, however, to understand which of these seem to matter the most we present results from several multivariate models in Table 2. All results are presented as standardized beta coefficients to compare the strength of associations across independent variables.

The strongest predictors of high ES dependency D_i, as shown in models I and II, are lower levels of education, less healthy, older, and lower socioeconomic status (as indicated by the asset index). Farmers and herders depend more on ES for their livelihood. Models III and IV on the other hand show factors associated with total ES value E_i. These models show much weaker relationships, with only marginal significance attributed to younger households and higher socioeconomic status. Herders seems to gain more from ES relative to the other household types. The data also explain much more of the variation in D_i (models I and II adjusted R^2 values are over 75%) compared to E_i (adjusted R^2 for models III and IV are both around 18%).

Putting these together, metrics of development and market integration, such as education levels and increases in assets are associated with less dependence on ES. Vulnerable households, such as those that are poorer, have a higher proportion of elderly, or with some chronic illness or injury, have greater dependence on ES. However, the total value of ES that households use does not seem to change as much with household demographic factors, although elderly households tend to use slightly lower values of ES, and ES utilization values still seem to increase with wealth.

5. Discussion

5.1. A consistent measure of ES flows to households

This paper develops measures of household-level ES value and dependence that can be consistently applied to populations in various locations. Such measures facilitate comparisons and better analysis of tradeoffs and impacts of ES change to various populations. The ES research and policy community have critically lacked ways to assess who might win or lose from landscape or policy changes, such as PES programs. The methods developed here provide several advantages for helping inform this kind of planning and management. First, these methods clearly delineate ES flow to human well-being from the landscapes' capacity to supply those ES by disaggregating ES flows into household-level livelihood inputs (Fig. 3) and who benefits from various ES (Fig. 5). This allows such changes to be documented over time or modeled for potential scenario analysis. Second, the method helps show how households access ES in different ways. Households with similar demographic characteristics engage in various livelihood activities, which have large implications for type and quantity of ES upon which they rely. Third, these methods provide important information that can help landscape managers develop tailored management strategies to support particular groups of interest (e.g., vulnerable or disadvantaged groups) (Fig. 5).

The differences in our results between D_i, the share of livelihood that comes from ES, and E_i, the total value of ES used by households, have several overarching implications. First, the positive relationship between ES value and non-ES values (Fig. 3d) suggests that safeguarding ES and investing in landscape protection has similar welfare impacts across all rural households, at least for those in our sample. While this support likely has an overall higher impact on poorer households (Fig. 3c), the fact that absolute value of ES to livelihoods seem to rise with wealth (Fig. 3d, Table 2 models III and IV) means that this matters for all households more than perhaps is traditionally

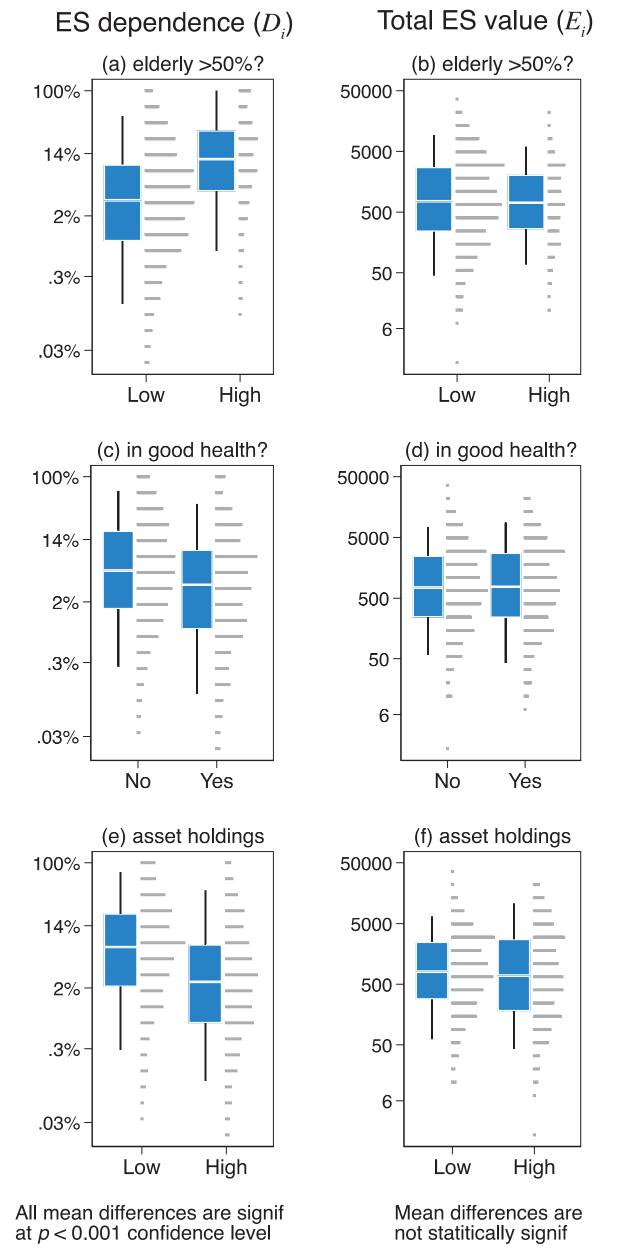

Fig. 5. Ecosystem service dependence varies by populations of interest. Disaggregating household-level ES benefits across subgroups of interest shows that, in our sample, households that are elderly-dominated (a and b), have a member with a chronic health condition (c and d), and have lower asset holdings (e and f) rely more on ES (mean differences are significant at $p < 0.001$ confidence level) (left column), but derive similar amounts of value from ES (right column).

assumed. Our data show that while use of ES grow with wealth, income shares from other sources grow more, making households with lower livelihood levels more heavily reliant on ES to get by.

5.2. Implications for land management

Measuring flows of ES at the household-level has several notable implications for practice and research. First, it could help catalog the impacts of landscape or policy change on local livelihoods, and better target poverty alleviation by safeguarding the most critical ecosystems. As we see above, ES policies that target households with lower socioeconomic status might preferentially address forest resources or help increase access to agricultural lands for these groups (Fig. 4b). On a practical level, most government and non-governmental organizations institutions require social and environmental assessments to understand impacts of proposed programs on various stakeholders, often specifically vulnerable and marginalized communities. These methods provide important distributional information that, alongside aggregate measures of efficiency that typical cost-benefit analyses provide, are often required for program and project approval. As an example, the United States' Millennium Challenge Corporation requires proposed projects to undergo a "Beneficiary Analysis" in addition to an aggregate calculation of a program's economic rate of return (MCC, 2018), which aims to "determine specifically *which segments* of society will benefit from the proposed activity" (pg. 1, emphasis their own).

Second, these methods could help support spatially explicit management as livelihoods, and thus ES values, vary across locations. For example, in village 18 tourism-related livelihood opportunities are important, thus a program that supports broad-scale landscape protection would likely help nurture and supported this industry (Fig. 3b). Alternatively, village 12 shows a relatively high reliance on ES to agriculture. Thus, encouraging best management practices that support ES inputs to agriculture, such as soil retention and on-site water management, could likely encompass a livelihood support strategy that could be broadly beneficial (Fig. 3b). Village 3 may gain more than other villages from a pest management interventions, as expenditures on pesticide seem to be a relatively large ecosystem *dis*service (Rasmussen et al., 2016) (Fig. 3b).

Third, measuring household ES values can also help improve specific management for targeted user groups. Other policies may target specific user groups across the region, as opposed to specific locations. For example, a region-wide policy to protect forest-based ecosystem services would disproportionately benefit households that are not farmers or herders, as the "other" household category (which has approximately 30% reliance on forests services – Fig. 4a) and those that are less wealthy (Fig. 4b). Similarly, the "other" household category seems to benefit more from current policy payments more than farmers and herders (Fig. 4a).

Finally, we see potential for better linking models of ES supply to measures of ES dependency to evaluate risk and uncertainty in the provision of ES. For example, climate change will impact changes the biophysical delivery and availability of ES. Measurements of ES dependence are needed to then understand which communities or populations are most vulnerable to those changes. Linking these to household-level factors that affect dependence, like those shown in Table 2, could help inform climate adaptation strategies. For example, Table 2 suggests that improving education and providing broad-based health insurance could help limit direct reliance on ES. Similarly, policy proposals with strong landscape dimensions, such as PES programs, could be modeled to identify resulting winners and losers.

6. Conclusions

This paper presents a standard method for disaggregating the value of ES to the household level. We focus on measuring the direct flow of ES to households, integrating ecosystem service valuation into a livelihoods framework. Our case study suggests disaggregating livelihood dependence on ecosystem services can provide novel and important land management information by helping reveal which aspects of the landscape are most important for supporting current livelihood conditions and identifying populations that are most at risk of landscape change. There may be various rationales for why households make livelihood choices that could range from personal preference, to local resource availability, to spillovers and teleconnections through markets or policy. These should be considered when designing particular strategies to impact livelihoods or ES dependence. As policy communities embrace an ecosystem services paradigm and the international

Table 2
Household factors associated with ES dependency and value.

Dependent variable	ES dependency: $\ln(D_i)$				ES value: $\ln(E_i)$			
	Model I		Model II		Model III		Model IV	
max education	−0.085***	(0.028)	−0.078***	(0.026)	−0.025	(0.010)	−0.02	(0.010)
healthy (0/1)	−0.050***	(0.069)	−0.042***	(0.066)	0.018	(0.026)	0.022	(0.026)
Elderly ratio	0.053***	(0.084)	0.069***	(0.081)	−0.075**	(0.032)	−0.063*	(0.032)
# kids in school	0.001	(0.055)	0.015	(0.053)	−0.016	(0.021)	−0.01	(0.021)
Loan amt in past 5 yrs	−0.005	(0.000)	−0.004	(0.000)	−0.021	(0.000)	−0.021	(0.000)
Asset index (wealth)	−0.086***	(0.044)	−0.073***	(0.042)	0.060*	(0.017)	0.061*	(0.017)
'*farmer*' (relative to '*other*')			0.153***	(0.093)			0.045	(0.037)
'*herder*' (relative to '*other*')			0.057***	(0.099)			0.117***	(0.039)
N	1749		1749		1749		1749	
Adjusted R^2	0.77		0.79		0.18		0.18	

Notes: Results are standardized beta coefficients from an ordinary least-squares model; standard errors are in parentheses. All models include village-level fixed effects and an indicator variable for households that have no estimated ES dependency (not shown).
* $p < 0.05$.
** $p < 0.01$.
*** $p < 0.001$.

community frames global goals, such as the Sustainable Development Goals (Griggs et al., 2013), around targets that rely on adequate provision of ES (Wood et al., 2018), it is necessary we better understand which populations benefit and which might suffer from future ecosystem, landscape, and related policy changes.

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