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Environmental incomes sustained as provisioning ecosystem service availability declines along a woodland resource gradient in Zimbabwe



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ABSTRACT

Forest and woodland resources can play a key role in rural livelihoods in the Global South, making it critical to understand what forest change could mean for rural wellbeing. Calculating environmental income has become a popular method of highlighting the importance of environmental resources in livelihoods, but few studies have quantified both provisioning ecosystem service availability and environmental income in the same landscape, or disaggregated environmental income by source land cover. This limits our ability to anticipate how forest change could impact rural livelihoods and could result in management interventions detrimental to vulnerable groups. The objective of this study was therefore to explore links between woodland cover, provisioning ecosystem service availability and household environmental income by applying a novel interdisciplinary approach in six villages on a gradient of woodland resource availability in Zimbabwe. We firstly use techniques from quantitative ethnobotany to score the species underpinning six locally important provisioning ecosystem services, and combine these scores with data from 80 tree survey plots to establish provisioning service availability. We then use income data from 91 households to explore relationships between provisioning service availability and household income portfolios. We find that villages with less woodland have lower availability of all studied ecosystem services and also a lower diversity of species underpinning service provision, but that there are no significant relationships between woodland resource availability and environmental income, livelihood diversity or intra-community income inequality in the case study area. We suggest that income portfolios are very resilient to woodland loss because households can still derive significant resources from woodlands which would be considered degraded in ecological terms and can draw upon kin networks which facilitate access to resources beyond village boundaries. The novel combination of approaches used in this study, particularly if applied at greater spatial and temporal scales, can provide valuable insight into the complexities of resource use in forest-agriculture mosaics.

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1. Introduction

Forest and woodland resources play a key role in rural livelihoods in developing countries (Byron & Arnold, 1999; Sunderlin et al., 2005). Over 2.7 billion people worldwide are estimated to depend on woodfuel for their primary energy needs (Bonjour et al., 2013), and woodfuel is just one component of an often under-recognised ‘hidden harvest’ (Campbell & Luckert, 2002) of environmental resources also comprising agricultural inputs, construction materials, medicinal plants and a high diversity of wild-

sourced foods (Campbell, Vermeulen, & Lynam, 1991; Ickowitz, Powell, Salim, & Sunderland, 2014; Powell et al., 2015; Rowland, Ickowitz, Powell, Nasi, & Sunderland, 2017; Galway, Acharya, & Jones, 2018). This so-called ‘environmental dependence’ of rural livelihoods in the Global South means that it is critical to understand the interactions between rural livelihood strategies and landscape structure, particularly in light of the widespread deforestation and degradation currently occurring in forest and woodland ecosystems.

The calculation of environmental income has gained prominence in recent years as a method of visualising the importance of non-cultivated resources in rural livelihood strategies, with a recent global analysis suggesting environmental resources to account for on average 28% of household income in rural areas of developing countries (Angelsen et al., 2014). Environmental

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income has been observed to enhance the diversity of rural livelihoods (Tesfaye, Roos, Campbell, & Bohlin, 2011), with diversification of income sources being an important pre-emptive adaptive strategy in regions characterised by high uncertainty (Ellis, 2000; Dercon, 2002; Debela, Shively, Angelsen, & Wik, 2012). Environmental income is also particularly important for very poor households who lack the capitals to access other livelihood options (Cavendish, 2000; Fisher, 2004; Mamo, Sjaastad, & Vedeld, 2007), and one consequence of this higher environmental dependence of poorer households is that inclusion of environmental income also reduces measured intra-community income inequality (Kamanga, Vedeld, & Sjaastad, 2009; Heubach, Wittig, Nuppenau, & Hahn, 2011; Kalaba, Quinn, & Dougill, 2013).

The logical progression from the above might therefore appear to be that communities which are otherwise similar but which differ in forest resource availability will also show differences in intra-community income inequality, and that the total income of households in communities with less access to forest will be lower and derived from a lower diversity of sources.

However, two major gaps in current literature limit our insight into the links between forest resource availability and household income portfolios. The first is that there are very few studies which quantify both resource availability and environmental income for the same study households. Studies have used distance to forest (e.g. Kamanga et al., 2009) or household forest land allocations (e.g. Hogarth, Belcher, Campbell, & Stacey, 2013) as proxies for access to forest resources, but these are imperfect metrics, because forests in the same landscape but with different ecological structure may support very different sets of provisioning ecosystem services (Woollen et al., 2016). The second is that only a minority of studies disaggregate environmental income by source land cover type. Many authors conflate 'forest income' with 'environmental income', or otherwise tacitly assume that forests have the highest concentration of livelihood-relevant resources and will thus be the primary source of environmental income (see e.g. Jumbe, Bwalya, & Husselman, 2008; Kalaba et al., 2013; Rowland et al., 2017), but this assumption is not universally supported in complex forest-agriculture mosaics (Dawson & Martin, 2015; Rasmussen et al., 2016; Zähringer, Schwilch, Andriamihaja, Ramamonjisoa, & Messerli, 2017). Angelsen et al. (2014) did find forest to be the dominant environmental income source in their global analysis, but this may be attributable to their use of the very broad FAO (2000) definition of 'forest'; several studies which have applied higher-resolution, locally derived land cover classifications have in contrast found that high biomass forests are not the primary source of environmental income, because they are subject to greater physical and social access barriers than other land cover types (Ambrose-Oji, 2003; Pouliot & Treue, 2013).

Addressing these literature gaps poses a challenge, as quantifying both resource availability and environmental income requires the combination of a diverse array of social and ecological research approaches. It should however be considered a priority for two reasons. Firstly, without studies quantifying both resource availability and patterns of actual use it is impossible to infer how forest change may impact household incomes or whether there are thresholds of forest resource availability at which households are forced to alter their livelihood strategies. Secondly, erroneous assumptions about patterns of resource collection based on preconceptions of which land covers are 'valuable' or 'degraded' can result in management interventions which are inadvertently harmful to vulnerable groups: McElwee (2009), for example, documents how the reforestation of land perceived as degraded in Vietnam resulted in the monopolisation of a previously common property resource by wealthier actors to the detriment of poorer households.

Our objective in this study was therefore to develop a new methodological approach quantifying both provisioning ecosystem service availability and household environmental income, and to apply this approach in six communities on a gradient of savanna woodland cover in Zimbabwe. Through this study we seek to explore the relationships between woodland cover, provisioning ecosystem services and rural livelihood strategies in Zimbabwe, and also to test a method which if applied at larger temporal and spatial scales could greatly improve our understanding of landscape structure-livelihood interactions.

2. Methods

Here we define key terms and establish our conceptual approach, before providing an overview of the study landscape. We outline the methods used to establish woodland cover and provisioning ecosystem service availability, including woodland surveys, focus groups and questionnaires quantifying the use value of woody species. We then detail the household income survey and the methods used to value non-market resources.

2.1. Conceptual background and key definitions

Environmental income is most often defined as all income from non-cultivated wild resources, thus including non-cultivated resources from agricultural land such as edible weedy plants (Sjaastad, Angelsen, Vedeld, & Bojö, 2005). One issue with this prevalent definition is that rural households in Zimbabwe also utilise numerous inorganic environmental resources such as sands for building and precious metals such as gold (Cavendish, 2000). We therefore follow Cavendish (2000) in including minerals under environmental income, but disaggregate the **organic** and **inorganic** components, **organic environmental income** corresponding to the definition developed by Sjaastad et al. (2005).

Household income in this study refers to **total net income**, the definition used in the majority of environmental income studies (see e.g. Cavendish, 2000; Mamo et al., 2007; Kamanga et al., 2009; Heubach et al., 2011; Angelsen et al., 2014). Total income includes all cash and subsistence income from agricultural products, livestock, employment, transfers and environmental resources, minus the value of inputs such as fertiliser and feed, and is seen as a more appropriate metric than cash income only because a large proportion of income in southern African communities is derived from own-produced or own-collected resources (Cavendish, 2000). Also in common with previous studies, the value of own labour is not deducted from net income, as it is not possible to establish appropriate shadow prices for labour in areas without functioning labour markets (Campbell & Luckert, 2002).

The value of expressing income from environmental resources in monetary terms is that it permits direct comparison between environmental income and other elements of household livelihood strategies. A challenge is that household income is highly variable in rural Africa, with income fluctuations meaning that many households experience periods of transitory poverty (Baulch & Hoddinott, 2000). This will also be reflected in the **environmental dependence** of rural households, commonly defined as the proportion of net annual household income derived from environmental resources (Mamo et al., 2007; Kamanga et al., 2009). In the absence of long-term panel data, we believe that calculation of income portfolio composition over a single year still represents the best method of analysing linkages between landscape structure and livelihood strategies, but also address the implications of a single year timeframe for our findings in the discussion.

While the majority of the environmental income literature focuses upon forest resources and disaggregates 'forest' and 'non-forest' income, we have chosen to avoid the term **forest** in

the majority of this paper. This may seem a surprising decision, but is motivated by a number of factors. Firstly, there is substantial debate over the definition of the term ‘forest’ (Chazdon et al., 2016), and in ecological terms the savanna systems which are the focus of this study are more commonly classed as **woodlands** (Frost, 1996; Chidumayo & Gumbo, 2010). Many environmental income studies avoid these definitional issues by using the FAO (2000) definition of forest (‘land spanning more than 0.5 ha with trees higher than 5 m and a canopy cover of more than 10 percent, or trees able to reach these thresholds in situ’), but we strongly believe that the breadth of this definition obscures important variation in the ecological characteristics of and social controls on different woodland areas, and that using this definition would have given us only limited understanding of the ways in which rural Zimbabweans source environmental resources. Throughout this research project we also aimed to see the landscape from the viewpoint of local people, as we believe this to have the greatest value for informing future management strategies, and this meant that use of a locally derived land cover typology was more appropriate – this ‘bottom-up’ approach is also embodied in our use of ethnobotanical methods to identify the woody species underpinning ecosystem service provision. We do however recognise that the choice of land cover categorisation has very important ramifications for our findings, and so consider how use of the FAO (2000) definition may have changed the outcome of this study in our discussion. While we ourselves choose to use the term woodlands, we believe that the methods described here could be applied in any tropical mosaic landscape, and that our findings have implications for the broader environmental and forest income literature.

2.2. Case study landscape: Wedza Communal Area, Mashonaland East, Zimbabwe

The focal landscape for this study is Wedza Communal Area, situated on and around Wedza Mountain in the Mashonaland East province of Zimbabwe. This area was chosen as the subject of study because a mixture of traditional beliefs, poor suitability for cultivation and past legal restrictions have resulted in Wedza Mountain maintaining cover of high biomass woodlands (Gumbo, 1988), while woodlands in the surrounding lowlands have been increasingly cleared or persist in an ecologically degraded state. Villages at varying distances from the mountain therefore lie on a gradient of access to woodland resources. Woodland in the study area is of the dry miombo type, characterised by dominance of *Brachystegia spiciformis*, *B. boehmii* and *Julbernardia globiflora* (Frost, 1996).

The issue of links between landscape structure and household income is particularly pertinent to the Miombo Ecoregion of southern Africa. Rural households in southern Africa derive a wide diversity of ecosystem services from woodland systems (Deweese et al., 2010; Ryan et al., 2016) and environmental income constitutes a substantial proportion of household income portfolios (Cavendish, 2000; Kalaba et al., 2013; Ryan et al., 2016). However, human activity is having widespread impacts on the structure and extent of miombo savanna woodlands (Ahrends et al., 2010; Jew, Dougill, Sallu, O’Connell, & Benton, 2016; McNicol, Ryan, & Mitchard, 2018), and the identification of the region as having high potential for forest landscape restoration (World Resources Institute, 2014) may drive further landscape transformation (a subject of considerable controversy; see Veldman et al., 2015a,b; Bond, 2016). Understanding of the land covers and resources currently important in household income portfolios is therefore important to support the design of landscape management interventions which will not detrimentally impact rural livelihoods.

Following a pilot visit in April 2014, three pairs of adjacent villages were chosen for inclusion in the study (Fig. 1): (1) Makumbe and Pfende in the deforested lowlands to the west of the mountain,

adjacent to the tarred road to Harare; (2) Mapfanya and Betera on the western edge of the mountain woodland; and (3) Mbizi and Charambira on the eastern side of the mountain between the woodland and the Sabi River. Longer-term residents of Makumbe and Pfende suggest that a substantial proportion of woodland clearance has occurred in the last 40–50 years. While initially selected based on observed differences in woodland cover, these villages also differ in terms of market access: Makumbe and Pfende have good transport links to local growth points and are only two hours from Harare by public transport, whereas Mbizi and Charambira are more remote and have poorer transport links. There are also important differences in access to mineral resources, with Mapfanya and Mbizi residents having claims to gold mining concessions on Wedza Mountain, and in water resources, with no major non-seasonal watercourses in Mapfanya and Betera.

2.3. Characterising woodland cover and provisioning ecosystem service availability

2.3.1. Village maps and land cover typology

The first phase of characterising ecosystem service availability was to create land cover maps for each of the six study villages in order to establish levels of woodland cover. A local land cover typology (Table 1) was developed based on data from two participatory mapping groups and four transect walks with key informants in each of the six study villages in April/May 2014. These data were combined with Google Earth satellite imagery using Development Team (2016). The resulting land cover maps (Fig. 1) were subsequently checked for accuracy with focus groups of purposively sampled key informants in June 2014. These focus groups also helped create the sampling frame for the household questionnaire, providing the gender and name of the household head for each village household, the number of adults and children in the household, and whether households were permanent residents in the study village (a number of households live in the city and keep the rural home only as a contingency).

2.3.2. Characterising village woodland cover

The composition and abundance of woody plants was assessed using survey plots. Five plot locations were randomly generated in QGIS for each village in the three most widely occurring land cover categories (low disturbance mountain woodland, high disturbance lowland woodland and fields), resulting in a total of 80 plots (as the two lowland villages have no mountain woodland area). Survey plots were circular with a radius of 20 m. Diameter at Breast Height (DBH: measured at 1.3 m) and local vernacular (Shona) name were recorded for all stems with DBH ≥ 3 cm. Local vernacular names were converted to scientific names using Mullin (2006) and Hyde, Wursten, Ballings, and Coates Palgrave (2016) and identification confirmed using Coates Palgrave (2002). Samples of species which could not be identified in the field were taken to the National Herbarium of Zimbabwe in Harare. DBH data were converted to biomass in dry matter (DM) using the mean of three allometric equations derived from similar dry Miombo systems (Grundy, 1995; Chidumayo, 1997; Ryan, Williams, & Grace, 2011). The woodland resource available to each household was calculated by dividing the estimated total woody biomass in the village area by the number of inhabited households in the village.

2.3.3. Quantifying provisioning service availability

Six wild-sourced provisioning ecosystem services were identified as particularly important based on the household survey (described below): firewood, construction materials, fibres, wild foods, medicinal plants and leaf litter fertiliser. Given our interest in the values of trees in landscapes, we focus this part of our

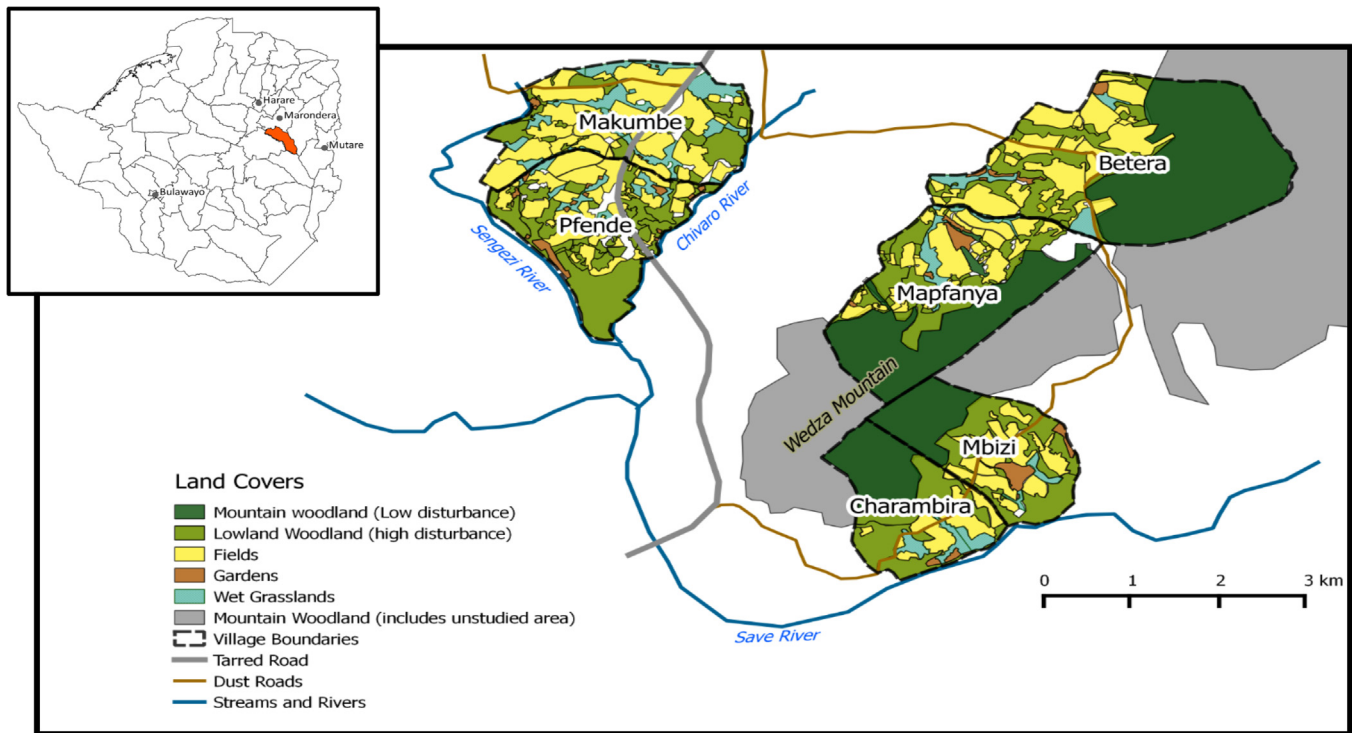


Fig. 1. Location of Wedza District (orange) with respect to major population centres in Zimbabwe, and the location and land cover patterns of the six study villages around Wedza Mountain. A local land cover typology was derived through participatory mapping and transect walks, and subsequently combined with imagery from Google Earth in Development Team (2016) to produce the maps above. The boundaries of the less-disturbed high biomass mountain woodland initially identified through the village-level data collection were triangulated using the Hansen et al. (2013) maps of global forest cover. The white intervening area includes additional villages also containing more disturbed lowland woodland areas, but these villages were not included in the research project and we therefore do not have the participatory mapping data required for their inclusion in the figure. Households are scattered throughout the village area, primarily concentrated in areas covered by fields. (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

Table 1
Land cover typology for Wedza Communal Area, Zimbabwe, created based on participatory mapping and transect walks with key informants. Mountain woodland, grazing area woodland and riparian woodland would all be captured within the FAO (2000) forest definition. This definition would also include some of the larger kopjes, but the majority of kopjes and termitaria would fall below the 0.5 ha threshold.

Land Cover Category	Sub-Category	Definition
Mountain Woodland (low disturbance)		Comparatively undisturbed high biomass miombo woodland found on Wedza Mountain.
Lowland Woodland (high disturbance)	Grazing area woodlands and abandoned fields	More disturbed lowland woodlands, found in village grazing areas. Also long-term abandoned fields with significant tree re-growth
	Kopjes/Termitaria	Small rocky hills (kopjes or zvikomo) and large termitaria which often retain vegetation cover when surrounding areas are cleared for agriculture
	Riparian woodland	Remnant woodland on river and streambanks
Wet grasslands		Seasonally dry (locally termed bani) and non-seasonal (dofonya) wetlands
Streams/ivers		
Fields/Gardens		Active/recently fallowed fields (including field margins)
Gardens		Small plots near to water sources used for cultivating vegetables
Built areas		Areas without vegetation cover, such as roads, household yards and borrow pits left following road construction

analysis only on the woody species which contribute to provisioning services.

Tree use values were determined using techniques from quantitative ethnobotany (Phillips & Gentry, 1993a,b) and closely followed those used by Luoga, Witkowski, and Balkwill (2000) in Tanzanian miombo. 87 woody ethnosppecies were identified from woodland survey data, the term *ethnosppecies* referring to locally recognised 'folk' species rather than scientific species. The full list was split into eight subsets of between 9 and 12 ethnosppecies, and the subsets were randomly assigned as a questionnaire module to an eighth of the 91 households involved in the environmen-

tal income survey. Four key informants identified as particularly knowledgeable during earlier field seasons answered questions about half of the full ethnosppecies list, and two local traditional healers discussed the full list of 87 ethnosppecies.

Questions on tree uses were targeted at the member of the household identified by the family as having the best ethnobotanical knowledge. Following Kristensen and Lykke (2003), respondents were first asked if they recognised the name of the tree. If they confirmed recognition, they were then asked whether the tree was useful as firewood, construction material, fibre, food, medicine or fertiliser. Trees were assigned a score of 0 (not useful),

Table 2
Metrics used to assess availability of six provisioning ecosystem services derived from woody ethnosppecies in villages around Wedza Mountain, Zimbabwe.

Provisioning Service	Availability Metric
Firewood	Dry mass of ethnosppecies scoring ≥ 1 in firewood category ¹
Construction poles	Number of stems of ethnosppecies scoring ≥ 1 for construction with DBH of ≥ 6 cm, based on measurements of poles used by rural households in Grundy et al. (1993)
Fibre	Number of stems of ethnosppecies scoring ≥ 1 as fibre with DBH ≤ 6.1 cm, as fibre is preferentially removed from small stems and 90% of stems observed stripped of fibre in woodland surveys were below the 6.1 cm threshold
Food	Number of stems of ethnosppecies scoring ≥ 1 as food. Quality of food availability was assessed by determining diversity of fruiting species and seasonal coverage of fruit availability (identified using fruiting dates reported in Coates Palgrave, 2002)
Medicinal Plants	Number of stems scoring ≥ 1 as medicine. Quality of medicinal plant availability was determined by coding qualitative responses to identify the number of conditions with locally available treatments and the number of potential remedies available for each type of condition. We report all locally perceived uses, making no judgement on the pharmaceutical validity of local knowledge
Leaf litter fertiliser	Annual leaf production from all stems scoring ≥ 1 as fertiliser estimated using allometric equations from Chidumayo (1997). As Miombo woodlands are deciduous, annual leaf production was assumed to be equal to annual leaf litter production

¹ A score of 1 or more indicates that the majority of respondents considered the ethnosppecies to be at least moderately useful for that provisioning ecosystem service.

1 (moderately useful) or 2 (very useful). In total the tree use survey resulted in a mean of 14.0 ± 0.3 (SE) responses per ethnosppecies. Only seven ethnosppecies received less than ten responses, and these species accounted for only 1.4% of all measured stems. The mean of all responses for each ethnosppecies was calculated to give a 'usefulness' score of between 0 and 2 for each service.

Provisioning service availability at hectare, village and household scales was calculated by combining data on the woody species composition and spatial extent of different land cover types with metrics appropriate to each individual provisioning service (Table 2).

2.3.4. Establishing local perceptions of past ecosystem service change and current availability

One focus group was held in each of the six study villages in February 2015, each group consisting of between 4 and 6 purposively sampled key informants who had lived in the study area for over 20 years. Groups began with general discussion of changes in land cover and local farming practices. Respondents were then given a list of provisioning ecosystem services, identified as locally important through the household income survey, and were asked to class each service as 'increasing', 'decreasing' or 'no change'. Additional qualitative data were collected on the reasoning behind the placement of each service and thematically analysed using the charting method described in Laws, Harper, and Marcus (2003).

2.4. Characterising household income portfolios

2.4.1. Household selection and definition

The primary household data collection instrument used in this study was a household income questionnaire survey adapted from CIFOR-PEN (2008) and carried out three times between June 2014 and November 2015. 100 households were initially selected for inclusion in the study using stratified random sampling, the working definition of **household** in this study being a group of people living under the same roof and pooling resources (following CIFOR-PEN, 2007; Woollen et al., 2016). Household lists were stratified by household size (1–2 residents, 3–5 residents and 6+ residents) and gender of household head (male-headed, headed by widow or divorcee, or *de facto* female headed with husband working outside the study area more than six months of year; categories follow Cavendish, 2000). Households were randomly selected from stratified lists in proportion to village size and to the representation of each group within the village. Following discussion of household samples with village heads, a further 4 households were purposively added to represent livelihood strategies or wealth groups which were perceived as missing within the random sample. Survey attrition over the rounds of the questionnaire was

higher than expected from previous studies, suggested by local residents to reflect high population mobility due to Zimbabwe's economic situation, with the initial 104 households reduced to a sample of 91 households across the six villages. Village size ranged from 10 to 53 permanently inhabited households (mean of 33 households) and final sampling intensity ranged from 37 to 80%.

2.4.2. Household survey design and delivery

Rounds of the questionnaire survey were carried out in June/July 2014, February/March 2015 and October/November 2015, and recorded income from environmental resources (both organic and inorganic), agriculture (field crops, garden crops, livestock and livestock products), off-farm formal and informal employment, and transfers (remittances, pensions, rental incomes and government support). Recall periods were six months for large, irregular income sources such as livestock sales, remittances and field crops, and one month for smaller incomes such as those from garden vegetables and environmental resources. Reported incomes were combined to represent a full year from September/October 2014 to September/October 2015. Full details of the methods used to calculate income in all categories are provided in the [supplementary information \(SI Table 1\)](#).

Appointments were made with households in advance, and the survey carried out with the available adult most knowledgeable about household incomes. In the majority of cases this was the oldest woman in the household, as men were more likely to be engaged in off farm work. Surveys lasted between 45 min and 2 h. All surveys were carried out by the same two researchers working together, with an experienced research assistant who is also a resident of the study area asking the questions and translating responses, and the first author transcribing responses. This strategy limited the feasible sample size, but improved the consistency of data collection and allowed responses to be clarified and follow up questions to be asked during the interview. The familiarity of both interviewers with the local area (the first author was resident in Wedza Communal Area for 11 months in total between April 2014 and December 2015) also meant that it was possible to identify through discussion where environmental resources were derived, and to assign each source location to a land cover type using the village maps described above. This additionally enabled us to estimate the proportion of environmental income derived by the household from outside the boundaries of their home village. The only resources which could not be reliably assigned to a source land cover type were those collected by household members other than the main respondent when travelling between the household and another location, such as children collecting wild fruits when herding cattle or going to school.

In order to understand how household socioeconomic characteristics influence livelihood strategies, data were collected during the household survey on the gender and age of all household residents, as well as on household assets and structures identified as important during twelve key informant interviews discussing local wealth indicators. The wealth index developed from these key informant interviews (details provided in [SI Table 2](#)) was also used to assign each household a wealth index score. Resource demands are not equal for all household residents, and so we follow [Cavendish \(2000\)](#) in expressing all income per adult equivalent unit (aeu), where the first adult is assigned a score of 1, all additional household residents of 15 and over a score of 0.5, and all residents of 14 and under a score of 0.3 (the modified OECD equivalence scale first suggested by [Hargenaars, de Vos, & Zaidi, 1994](#)). Household members were counted as residents if they lived in the household for more than 6 months of the year. The dependency ratio was also calculated for each household, meaning the number of residents in the household = <14 and >66 years old relative to adults of working age. The distance in metres from the household to the nearest tarred road was also calculated using Google Earth, as this provides an indication of the ease with which households can access local and regional markets.

2.4.3. Valuing environmental resources

Valuation methods for all resources (including environmental resources, crops and livestock/livestock products) were based on [Cavendish \(2000, 2002\)](#), [Heubach et al. \(2011\)](#) and [Wunder, Luckert, and Smith-Hall \(2011\)](#). Where possible, the prices used were 'revealed' by transactions reported during the household survey or observed in the study area. For those resources without active local markets, respondents were asked to estimate willingness to pay (WTP) for the resource, or in the case of materials such as gravel their willingness to pay for the labour involved in resource collection (following [Heubach et al., 2011](#)). Respondents gave WTP either in cash terms or using a commonly bartered substitute with a well-recognised local price such as buckets of maize or bars of soap. In many cases WTP estimates clustered around a mean point, suggesting a consistent locally perceived monetary value ([Cavendish, 2002](#)). Where WTP estimates did not have a central tendency prices were imputed from similar resources: for example, widely differing estimates were given for the value of non-traded wild fruits such as **matufu** (*Vangueriopsis lanciflora*) and **matohwe** (*Azanza garckeana*), and so prices were instead inferred from locally traded fruits such as guava (*Psidium guajava*), **mazhanje** (*Uapaca kirkiana*) and **mut-subvu** (*Vitex payos*).

2.4.4. Analysis of household income data and livelihood diversity

Relationships between household income/livelihood diversification and household socioeconomic characteristics were analysed using multiple regression, with variables log transformed to meet regression assumptions as needed. Following [Ellis \(2000\)](#) and [Tsfaye et al. \(2011\)](#), the diversity of income sources and of environmental income sources was calculated using the inverse Simpson index of diversity, also termed the inverse Herfindahl-Hirschman index. This diversity index cannot be calculated with negative values and a number of households had net negative income in one or more categories (particularly input-heavy categories such as field crops), and diversity indices were therefore calculated both using gross income in each category and using net total income with all negative values converted to zero. Following numerous studies of income inequality (e.g. [Kamanga et al., 2009](#); [Kalaba et al., 2013](#)), the Gini coefficient was used to assess income inequality with and without inclusion of environmental income. All analyses were carried out in Excel and R ([R Core Team, 2014](#)).

3. Results

3.1. Woodland cover and provisioning ecosystem service availability

3.1.1. Village woodland resource availability

Relative village woodland cover (including both high and low disturbance woodland as outlined in [Table 1](#)) varied from 19.4% in Makumbe village to 70.1% in Charambira village ([Table 3](#)). Pooling plot data from all villages, aboveground woody biomass was higher in less disturbed mountain woodland (44 ± 4 t dry matter (DM) ha^{-1} ; unless otherwise stated \pm refers to one standard error) than in more disturbed lowland woodland (15 ± 1 t DM ha^{-1}) or agricultural land (5 ± 1 t DM ha^{-1}). The estimated total woody biomass at the village scale is highest for Betera village ($13,920 \pm 492$ t DM) and lowest in Makumbe village (1070 ± 319 t DM). When woody resource is expressed per inhabited household, households in Charambira village have the highest level of woodland resource (658 ± 29 t DM hh^{-1}) and households in Makumbe the lowest (19 ± 6 t DM hh^{-1}).

3.1.2. Local perceptions of provisioning ecosystem service change

Woodland cover was reported to have decreased in recent decades by focus groups in all six study villages. Increase in local population was seen as an important proximate driver due to corresponding increases in demand for woody resources and agricultural land, and this was suggested to have been exacerbated by increased urban to rural migration during the economic crises of the 1990s and 2000s. Uncontrolled fires, wood harvest for tobacco curing, and reduced adherence to traditional taboos on woody species harvesting were also highlighted as causes of woodland loss. Groups also suggested that loss of tree cover, in combination with overgrazing and tillage of riparian land, has resulted in siltation of local watercourses.

Of the 15 provisioning services discussed with focus groups, nine were perceived to have declined in availability in four or more of the study villages ([Table 4](#)). Overharvest by incomers from other villages was a commonly cited cause of decline across multiple services, while declines in thatching grass were also attributed to overgrazing and declines in fish abundance to river siltation. In contrast, the thorn trees used for fencing vegetable gardens were perceived to have increased because they flourish in deforested areas. Bark fibres used for thatching and tying firewood headloads were also perceived to have increased in several villages because they are easier to harvest from the coppice stems which regrow following cutting of locally dominant *Brachystegia* spp. rather than from mature trees. Medicinal plant availability was not assigned any consistent trend, with some medicinal species increasing and others decreasing. However, respondents caveated reported trends for both medicinal plants and wild vegetables by saying that perceived declines might be indicative of changes in knowledge rather than abundance, with medicinal plant use discouraged by the increasingly influential Apostolic and Pentecostal churches, and with households preferring vegetables from fields and gardens to those in woodlands and wetlands.

3.1.3. Observed variation in provisioning ecosystem service availability

Data from the tree use survey show that firewood and construction poles are services which are supported by a large variety of ethnospecies, with 69 of the recorded ethnospecies being scored as at least moderately useful in each category. Leaf litter fertiliser could also be derived from a wide range of ethnospecies, with 52 ethnospecies classified as useful. The other three services depended on more specific subsets of ethnospecies; 26 ethnospecies scored as useful for food, 3 as useful for fibre, and 14 useful for medicine. However, it should be noted that medicinal plant knowledge was highly heterogeneous, and that all but one

Table 3Woodland cover and biomass of woody species in six villages around Wedza Mountain, Zimbabwe. \pm represents one standard error.

Position of village pair	Village	Woodland cover (% of total village area)	Estimated woodland biomass (t DM)	Woodland biomass per inhabited household (t DM hh ⁻¹)
Lowland villages	Makumbe	19.4	1070 \pm 319	19 \pm 6
	Pfende	48.2	1597 \pm 636	51 \pm 21
Western side of mountain	Mapfanya	67.6	10911 \pm 545	287 \pm 14
	Betera	69.4	13920 \pm 492	324 \pm 11
Eastern side of mountain	Charambira	70.1	6581 \pm 288	658 \pm 29
	Mbizi	65.6	5738 \pm 479	287 \pm 24

Table 4

Locally perceived directions of change in provisioning ecosystem service availability in six villages around Wedza Mountain, Zimbabwe (+ indicates an increase in availability, – a decrease in availability). Villages are ordered from lowest per household woody resource availability (Makumbe, left) to highest per household woody resource availability (Charambira, right).

Resource	Makumbe	Pfende	Mbizi	Mapfanya	Betera	Charambira
Firewood	–	–	–	–	–	–
Poles	–	–	–	–	–	–
Thatching grass	–	–	–	–	–	–
Thorn Trees	–	+	+	+	–	+
Fibres	–	–	+	–	+	+
Wild fruits	–	–	–	–	–	–
Wild vegetables	–	–	–	–	+	–
Bushmeat	–	–	–	No change	Not discussed	–
Fish	–	–	–	–	–	–
Ants/Termites	+	+	No change	+	No change	–
Caterpillars	Not discussed	–	No change	–	–	–
Locusts	–	+	No change	–	–	–
Mushrooms	–	No change	No change	–	+	–
Medicinal plants	No change	No change	–	–	+	Not discussed
Leaf Litter Fertiliser	–	–	+	–	No change	+

ethnospecies was attributed a medicinal use by at least one respondent. The overall highest scoring indigenous species were **muhacha** (*Parinari curatellifolia*), **mupfuti** (*Brachystegia boehmii*) and **munondo** (*Julbernardia globiflora*). Non-indigenous species such as mango (*Mangifera indica*) and guava (*Psidium guajava*) also scored highly, but respondents qualified that these trees are not common property resources in the same way as naturally occurring woodlands, instead being considered the property of the person who had planted them.

Declines in total village woodland resource were associated with declines in the availability of all studied services on a per household basis (Figure 2). The steepest decline in availability was in the standing stock of biomass suitable for firewood, which fell from 590 \pm 24 t DM hh⁻¹ in the village with highest overall woodland resource to 17 \pm 5 t DM hh⁻¹ in the village with lowest woodland resource. Number of fruit tree stems per household also declined steeply, from an estimated 5840 \pm 437 stems hh⁻¹ in the village with highest woodland resource to 94 \pm 64 stems hh⁻¹ in the village with lowest woodland resource. This decline in abundance was also associated with declines in the diversity of wild fruiting trees, with a lower number of species producing edible fruit in each month of the year in the two villages with lower woodland resource (Fig. 3). This lower richness of fruiting species is of particular concern during the ‘hungry gap’, the months prior to harvest when households begin to run out of the maize harvested the previous season (Hoddinott, 2006). The number of locally recognised categories of medical condition with a medicinal plant remedy within the village area was also lower in the villages with lower woodland resource, as was the number of potential medicinal plant remedies per category of condition (Fig. 4).

3.2. Composition and diversity of household income portfolios

3.2.1. Household characteristics

There were few apparent differences in the mean values of socioeconomic characteristics between household populations in

the six study villages. The mean age of household heads within the sample was 56 \pm 2 years. The mean household size across all households was 4.7 \pm 0.3 individuals, equating to 2.4 \pm 0.1 adult equivalent units (aeu). 53 households (58% of the whole sample) were headed by men, 31 (34%) by widowed or divorced women, and 7 (8%) by women with husbands working away from the rural area for more than 6 months of the year. The 13 households which did not complete all three rounds of the questionnaire were evenly distributed across the six villages, but had slightly higher representation of household heads over 70 years old and of women with husbands working away than the remaining sample.

3.2.2. Organic environmental income

Organic environmental income contributed the greatest proportion of household income portfolios, accounting for on average 31 \pm 2% of total net income. Garden crops, livestock and remittances were also important, the mean value for each being more than 10% of household income. Field crop income accounted for only 3.1 \pm 0.9% of total net income.

When comparing mean income portfolios for the six study villages, organic environmental income was the highest ranked income source in all villages (Fig. 5; absolute values in US\$ are provided in SI Table 3). Remittances and garden crops also ranked highly in all villages, and livestock made a high relative contribution in all villages except Makumbe. Conversely, skilled employment and pensions were highly ranked contributors only in Makumbe. Inorganic environmental income contributed to a higher proportion of total income portfolios in Mapfanya and Mbizi, the two villages with access claims to gold mining concessions.

Multiple regression analysis indicates that absolute organic environmental income is positively and significantly associated with total household income and negatively and significantly associated with household wealth index score, but was not significantly associated with any other household characteristics (all co-efficient estimates are provided in Table 5). The relative contribution of environmental income was negatively and

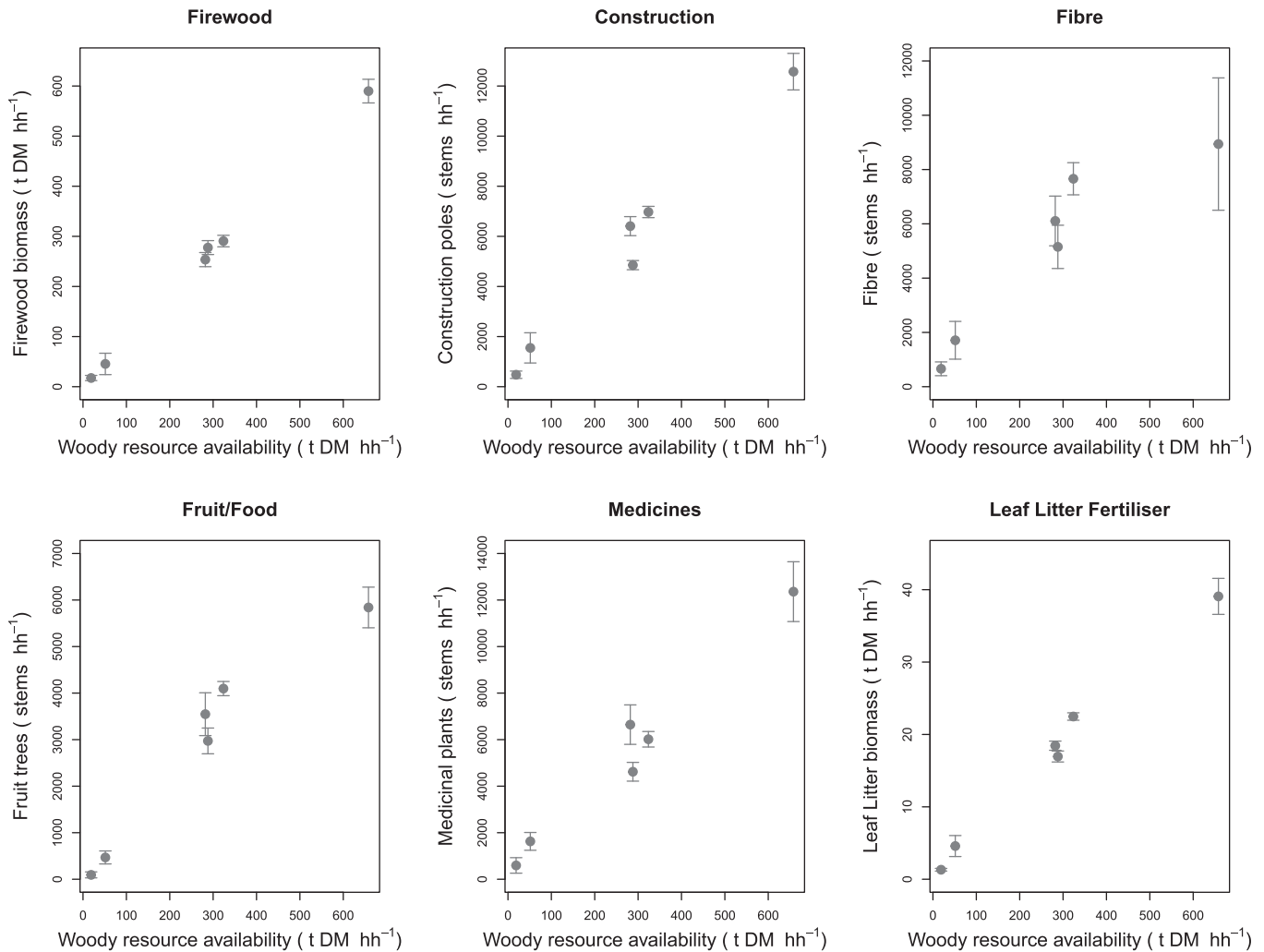


Fig. 2. Availability of six provisioning ecosystem services relative to the total woody resource available to each household in six villages around Wedza Mountain, Zimbabwe. Useful ethnospices in each category are those species scoring more than 1 (i.e. at least moderately useful) in a co-occurring survey on the uses of woody species, excepting medicinal plants where overall greater heterogeneity in knowledge and therefore lower mean scores made a usefulness threshold of 0.5 more appropriate. Error bars represent one standard error.

significantly associated with household wealth index score, but was not significantly associated with any other household characteristics. There was no significant relationship between the woody resource available to the household and either absolute organic environmental income or environmental dependence. It should however be noted that adjusted R^2 values were comparatively low for both the absolute organic environmental income and relative environmental income analyses (Table 5). There was initially a significant effect of village on organic environmental income (ANOVA: $F = 2.6$, $df = 5,85$, $p = 0.03$), with mean organic environmental income in Mbizi significantly higher than that in Makumbe (TukeyHSD: $p = 0.02$) but exclusion of a single outlier household in Mbizi village with environmental income three times that of the next highest household resulted in this relationship becoming non-significant (ANOVA: $F = 1.9$, $df = 5,84$, $p = 0.1$).

3.2.3. Composition and derivation of organic environmental income portfolios

Woodfuel was the most economically important environmental resource, accounting for on average $39 \pm 2\%$ of total household organic environmental income (equivalent to a value of US $\$106 \pm 7 \text{ aeu}^{-1} \text{ yr}^{-1}$). Wild foods including wild fruits, wild vegetables, fish, meat, mushrooms and insects were the second most

important source, accounting for on average $26 \pm 2\%$ of household organic environmental income. Construction materials such as thatching grass, poles and thorn trees contributed on average $22 \pm 2\%$ of household organic environmental income.

There was no significant difference between households in the six villages in the average absolute income derived from woodfuel, construction materials or wild-sourced fertilisers (Kruskal-Wallis tests: $p > 0.05$). There was a significant difference between villages in the mean average income derived from all wild food sources combined (Kruskal-Wallis: $\chi^2 = 14.4$, $df = 5$, $p = 0.01$), but this appears to be a consequence of the very high mean income from wild foods in Mbizi village rather than of lower wild food income in villages with lower woodland cover (a full breakdown of organic environmental income by source and village is provided in SI Table 4). Exclusion of Mbizi households from the analysis rendered the effect of village on wild food income non-significant (Kruskal-Wallis: $\chi^2 = 6.0$, $df = 4$, $p = 0.2$). The only income source which appeared to potentially co-vary with woodland cover was wild fruits, with households in the two villages with lowest woodland resource reporting mean income from wild fruits of US $\$23 \pm 7 \text{ aeu}^{-1} \text{ yr}^{-1}$ and US $\$23 \pm 11 \text{ aeu}^{-1} \text{ yr}^{-1}$, as opposed to between US $\$41 \pm 10 \text{ aeu}^{-1} \text{ yr}^{-1}$ and US $\$79 \pm 28 \text{ aeu}^{-1} \text{ yr}^{-1}$ in the villages with higher woodland resource. However, the

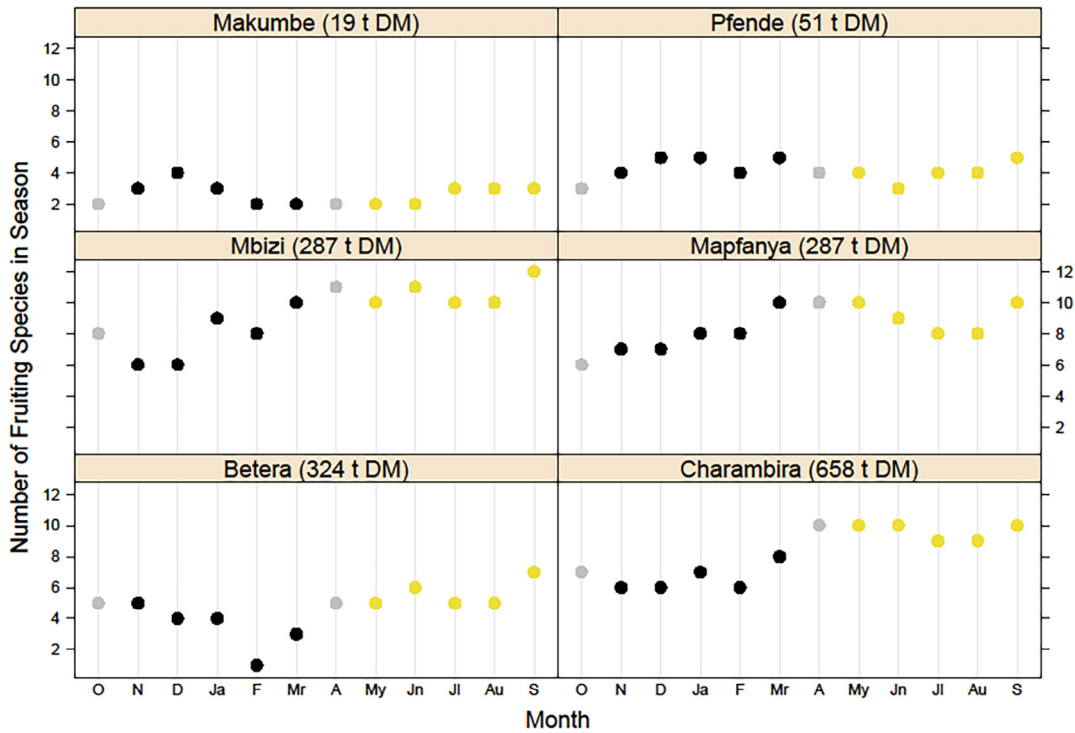


Fig. 3. Number of wild or naturalised ethnosppecies recorded in each village which produce edible fruit in each month of the year in six villages around Wedza Mountain, Zimbabwe. Numbers in brackets refer to the total woody biomass resource estimated to be available to each village household. Points between November and May (black points) fall within the ‘hungry season’, referring to the period of the year before the harvest when household stocks from the previous harvest are running low (Hoddinott, 2006). Note that this figure represents only indigenous or naturalised woody species, as the exotic fruit species planted in homestead yards are subject to different tree tenure rules and the fruit are not accessible to all village residents.

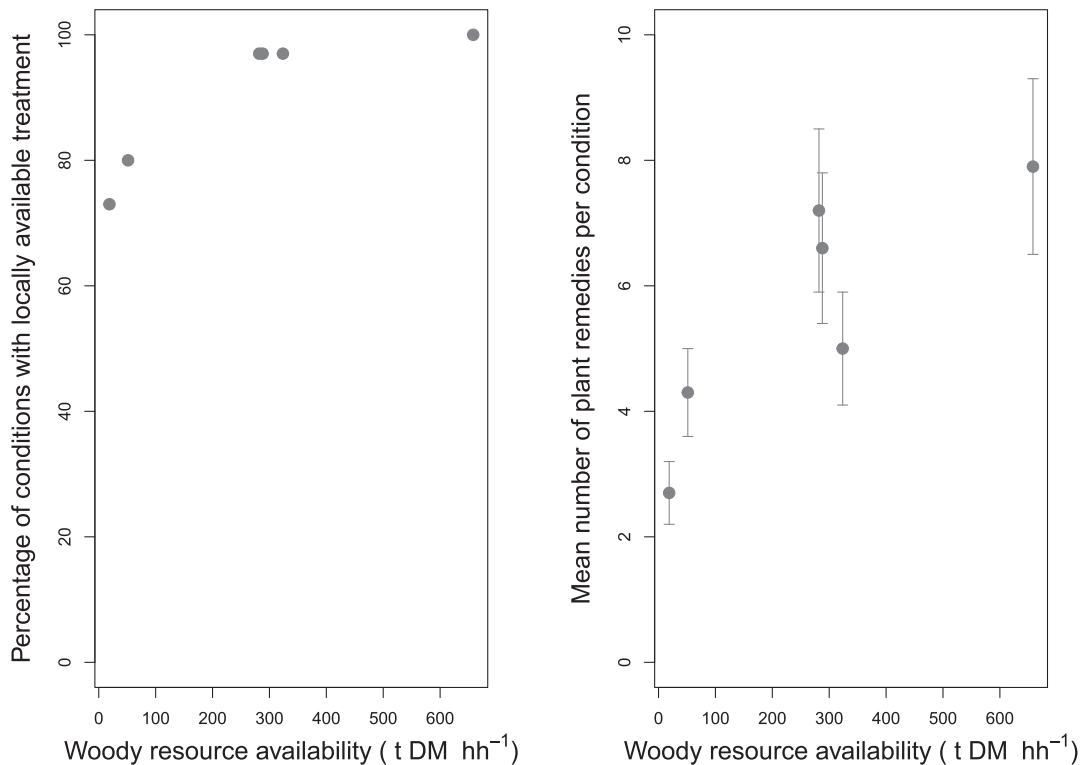


Fig. 4. Percentage of locally recognised categories of medical condition with a tree-derived remedy available in the village area (left), and the mean number of tree-derived remedies available within the village area per category of condition (right). Error bars represent one standard error.

relationship between village and wild fruit income was borderline in terms of statistical significance (Kruskal-Wallis: $\chi^2 = 10.5$, $df = 5$, $p = 0.06$).

Woodland ecosystems were the primary source of environmental income, accounting for on average $66 \pm 2\%$ of all reported organic environmental income. While less disturbed mountain

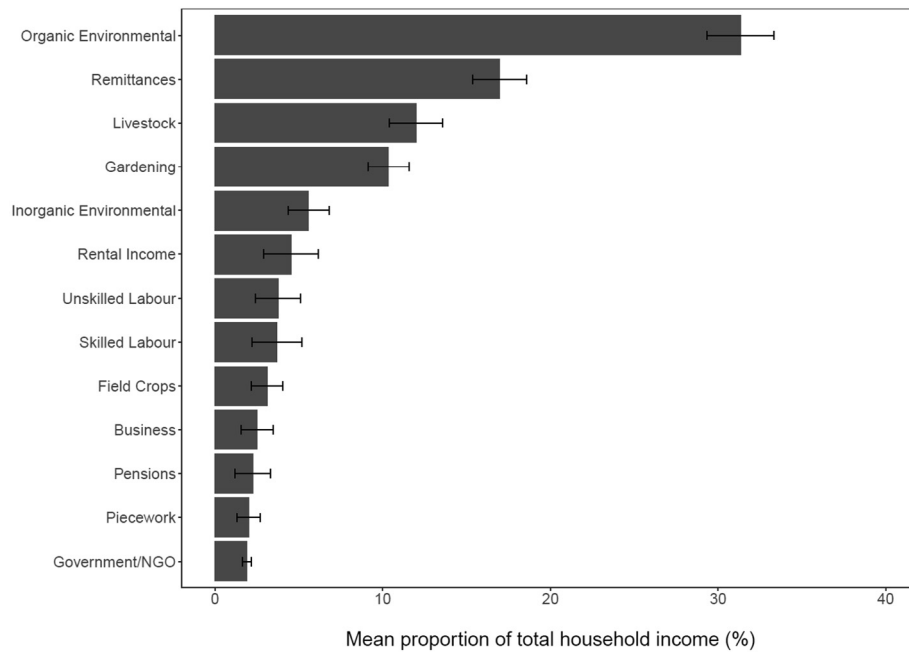


Fig. 5. Mean percentage contribution of all income sources to total net income for 91 households in Wedza Communal Area, Zimbabwe. Error bars represent one standard error.

Table 5
Multiple regression analysis of links between household characteristics, environmental income and environmental dependence for 91 households in Wedza District, Zimbabwe.

	Household Organic Environmental Income (log transformed: US\$ aeu ⁻¹ yr ⁻¹)		Household Environmental Dependence (log transformed: % of household income aeu ⁻¹ derived from organic environmental resources)	
	Co-efficient estimate (standard error)	P	Co-efficient estimate (standard error)	P
Total household income (log transformed: US\$ aeu ⁻¹ yr ⁻¹)	0.586	<0.001	-0.187	0.872
Household wealth index score	-0.020	<0.001	-0.296	<0.001
Woody resource availability (t DM hh ⁻¹)	0.000	0.603	0.000	0.584
Household head (female)	0.021	0.902	0.084	-0.644
Age of household head (years)	-0.003	0.678	0.003	0.679
Household size (log transformed: aeu)	0.226	0.320	0.047	0.064
Dependency ratio	-0.071	0.147	-0.052	0.430
Distance to tarred road (m)	0.000	0.1498	0.000	0.268
	<i>n</i> = 91; adjusted R ² = 26.3%		<i>n</i> = 91; adjusted R ² = 30.5%	

woodland was an important source of environmental income, accounting for between 23 ± 6% and 43 ± 6% of total organic environmental income in the mountain adjacent villages, a substantial proportion was also derived from more disturbed lowland grazing area and riparian woodland and from remnant woodland patches on kopjes and termitaria (Fig. 6). Fields and field margins were also an important source land cover, contributing 22 ± 2% of organic environmental income.

Income from poles and firewood was primarily derived from tree dominated land covers, with almost equal proportions derived from disturbed and less disturbed woodland systems. Wild fruits, in comparison, were primarily derived from more disturbed lowland woodlands. Wild vegetables and thatching grass were both predominantly derived from fields and field margins.

Respondents in Makumbe village reported a significantly higher proportion of organic environmental income having been obtained outside the village area (ANOVA: $F = 4.8$, $df = 5,85$, $p < 0.0001$), with households specifically identifying an average of 19 ± 5% of organic environmental income as having been obtained in land belonging to other villages. In contrast, only on average between 0 and 8% of household organic environmental income was identified as being obtained outside the community in the other

five villages. The resources most commonly reported as being collected outside the respondent's home village were firewood and wild fruits. However, this finding should be treated with some caution: the observed tendency of households was to collect resources as close to home as possible, but village boundaries are not clearly demarcated on the ground particularly in mountain woodland, and without directly tracking resource collection trips (either in person or using GPS technology) it is not possible to conclude with total certainty that a greater proportion of reported income is not obtained outside village boundaries.

3.2.4. Livelihood diversity and income inequality

The diversity of gross income sources (calculated using the inverse Simpson index) was positively and significantly associated with household wealth index score (coefficient = 0.373 ± 0.081 , $p < 0.0001$) and negatively and significantly associated with the log of household income (coefficient = -0.898 ± 0.188 , $p < 0.0001$), but was not significantly associated with woody resource availability or with household demographic characteristics (multiple regression: $n = 91$ households, adjusted R² = 0.28, relationship reported as significant if $p < 0.05$). The same significant relationships were

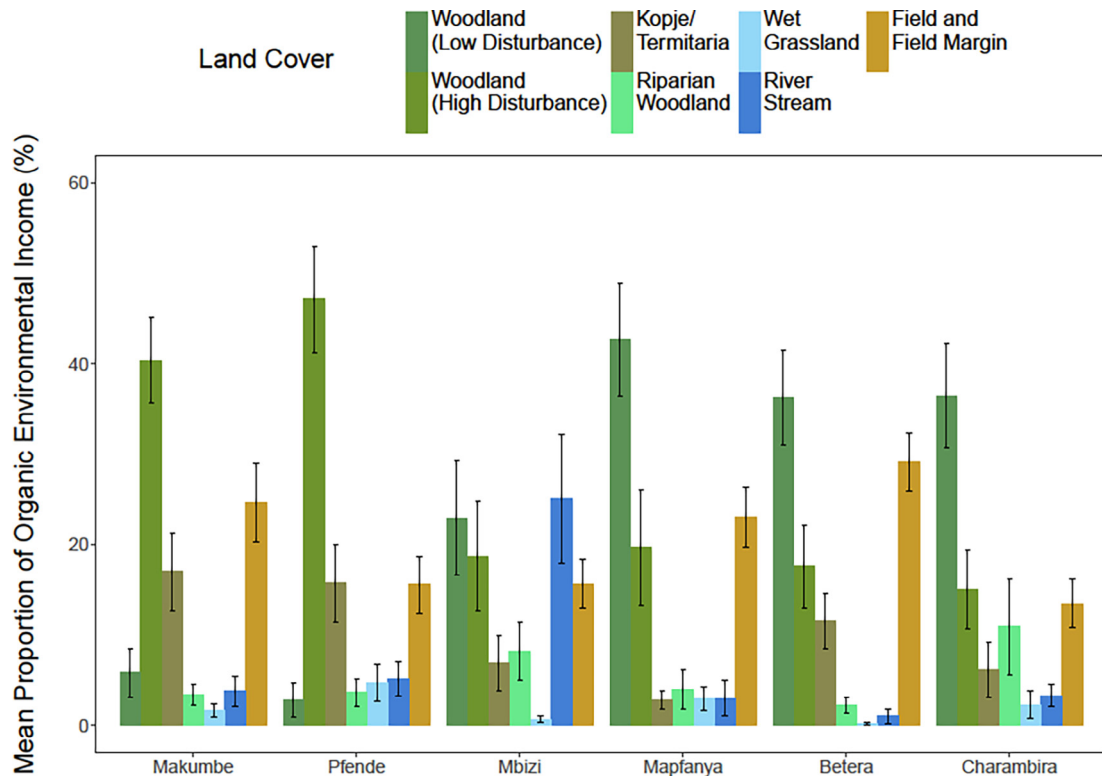


Fig. 6. Mean contribution of each land cover type to total organic environmental income. Error bars represent one standard error. Villages are ordered from lowest woody resource availability (Makumbe, left) to highest woody resource availability (Charambira, right). Plotted percentages do not equal 100% as the figure omits the proportion of organic environmental income derived from 'other' land cover types (e.g. homestead yards) and which could not be reliably assigned to a land cover type. The surprisingly high contribution of rivers and streams in Mbizi village is due to two households which were heavily dependent on fishing and on hunting for cane rats (*Thryonomys* sp.) on riverbanks.

observed when the analysis was performed based on the diversity of net total income with all negative values converted to zero.

The Gini coefficient of income inequality was 0.39 for the whole sample, ranging from 0.18 in Charambira to 0.5 in Makumbe. While inequality initially appears substantially higher in the village with lowest woodland resource, exclusion of a single outlier household with income seven times that of any other household in the sample meant that the Gini coefficient for Makumbe fell to 0.36. Exclusion of environmental income increased the Gini coefficient for the full sample to 0.47, and also resulted in increased income inequality in all study villages.

4. Discussion

4.1. Key finding: lower woodland cover is associated with reduced provisioning service availability but not with reduced environmental income

Our findings indicate a close association between woodland cover and provisioning service availability, with stocks of all six studied services lower in villages with lower woodland resource availability. These observations correspond closely with the declines in ecosystem services reported by village focus groups, and also parallel the results of Woollen et al. (2016), who found that degradation of miombo woodlands in Mozambique due to charcoal production can jeopardise the provision of other services. We also observe reductions in the diversity of ethnospecies underpinning the provision of ecosystem services in villages with lower woodland resource availability, raising concerns over the future resilience of these services to further changes in land use patterns and intensity.

However, these declines in provisioning services do not appear to translate to changes in the composition or diversity of household income portfolios in this context. We do not observe any difference in organic environmental income in villages with high and low woodland resource availability. Equally, while our results match those of previous studies (e.g. Kamanga et al., 2009; Dokken & Angelsen, 2015) in finding that less wealthy households have higher environmental dependence and that inclusion of environmental income reduces intra-community income inequality, we do not find higher income inequality in villages with lower provisioning service availability. Nor do we find any association between lower provisioning service availability and the diversity of rural livelihood strategies.

4.2. Why is environmental income not lower in villages with less woodland resources?

Our data suggest that one probable explanation for the lack of relationship between woodland resource availability and household environmental income lies in the ability of households to derive the environmental resources they need from non-woodland land covers and from woodlands which would be considered seriously degraded in purely ecological terms. Even in mountain adjacent villages, the average proportion of household environmental income derived from high-biomass mountain woodland was at most around 40% – a very substantial proportion, but demonstrating that high biomass forest and woodland systems cannot be conceptualised as 'islands of resources in otherwise barren landscapes. These findings also demonstrate the value of using locally-derived land cover categorisations rather than very broad 'top-down' definitions – while all the woodland categories and sub-categories in this study except for smaller kopjes and

termitaria would be included under the FAO (2000) forest definition used by studies such as Angelsen et al. (2014), reliance upon this definition alone would not have permitted us to illuminate the livelihood value of lower biomass systems and would have left our findings very open to misrepresentation as demonstrating a simple 'win-win' spatial congruence between ecological and livelihood values.

Patterns of environmental resource collection were driven in part by the habitats of preferred species, with thatching grass growing primarily on contour ridges between fields, and wild vegetables such as *Cleome gynandra* on the margins of maize fields. Others were primarily derived from more disturbed systems close to homesteads because they were collected incidentally to other livelihood activities, such as the wild fruits collected while herding cattle. Our observations in the case of firewood also lead us to suggest that barriers restricting the access of the main collectors (almost always the women in the household) to high biomass woodland, such as limited time availability around other caring commitments and lower claims to the use of scotch carts, may result in preferential collection of lower quality but local firewood ethnospices over those which are high quality but more distant from the household. These findings that the non-wooded land and degraded systems are important to rural livelihoods, and that many 'forest resources' can be substituted by resources from systems with less woody biomass, echo the findings of previous research in Africa (McGregor, 1995; Ambrose-Oji, 2003; Pouliot & Treue, 2013; Dawson & Martin, 2015) and suggest that household incomes may only be impacted at very high levels of woodland degradation.

The higher proportion of organic environmental income specifically identified as being derived from outside the village area in the village with lowest woodland cover indicates that an additional important coping mechanism for low local resource availability is to draw upon kin and social networks which facilitate access to resources in other communities. Many respondents in Makumbe travelled to the grazing area in Pfende village to obtain firewood, while others capitalised upon connections in the more wooded resettlement areas across the Sengezi River. Coping with resource limitations by travelling to other villages was also a strategy observed by Mandondo (2001), but can result in discord; residents of Pfende see use of their grazing area by interlopers from more deforested villages as a primary cause of woodland degradation. Also important to note is that very few households in Makumbe travelled to the mountain woodland to collect resources, instead drawing upon more local lower-biomass woodland areas, and so the derivation of organic environmental income from outside the study village does not undermine our conclusion that lower-biomass woodlands are extremely important to rural livelihoods.

The observation that residents of deforested villages have to travel further to obtain resources also suggests a need to look beyond environmental income alone, with income being only one aspect of multidimensional wellbeing (Vollmer et al., 2017). Travelling to other villages for resources relies upon access to appropriate cross-boundary social networks and often necessitates use of a scotch cart, which may render this strategy inaccessible to less wealthy households and so magnify resource access inequalities. Increased distance to resources can also result in reallocation of time from other productive activities and alter gendered resource collection roles, with as-yet underexplored consequences from intra-household variation in wellbeing (Cooke, Köhlin, & Hyde, 2008). Relatedly, the observation of a borderline significant relationship between wild fruit consumption and woodland resource availability hints that nutritional metrics may have greater sensitivity to changes in woodland cover (as discussed in Ickowitz et al., 2014). It is not however possible to confirm from our data whether this is a consequence of the lower diversity of fruiting spe-

cies recorded in the most deforested villages, or whether the shorter distance to markets in villages with lower woodland cover means that households are able to purchase alternative foods. While our results provide useful insights, there is thus much to be gained from studies which consider a greater range of wellbeing dimensions.

4.3. Caveats and next steps

As one of very few studies to quantify both provisioning service availability and environmental resource use for the same households, there are a number of study limitations which we believe it important to properly discuss and which we hope will motivate further research.

The first limitation is the single-year time frame used in this study. The high environmental dependence observed in this study may be partially attributable to the fact that research was carried out in a drought year, also explaining why field crop income accounts for on average only 3.1% of total income. Recall of previous harvests suggested that mean household maize production in 2015 was on average a little less than half that in 2013 and 2014, equivalent to a difference in gross income of around US \$190 (Pritchard, unpublished data). However, field crop production also requires the greatest amount of inputs of all income categories, with an average value of US\$234 hh^{-1} recorded in 2015 (the higher value than similar studies perhaps reflecting that inputs of livestock manure were in this case accounted for under livestock products). When income portfolio results were shared during feedback groups in April 2017 respondents further confirmed that high input intensity means that field crop production is often a loss-making exercise in income terms even in average years. Our estimate that organic environmental income accounts for on average 31% of total household income is also consistent with the 37% reported by Cavendish (2000) in Zimbabwe and the 32% observed by Angelsen et al. (2014) in sub-Saharan African study sites, giving us greater confidence that our income portfolio results are robust for the year in question. However, there is a major need for longitudinal panel data studies which assess the extent to which environmental income varies year on year. This would also enable analysis of changes in forest-livelihood interactions over time, which would be an improvement on current approaches such as the present study which are based upon space-for-time substitution (see also Woollen et al., 2016).

The second limitation is that the six study villages all lie within relatively close proximity, and this does mean that even mainly deforested villages were not entirely without woodland, as demonstrated by the finding that households in Makumbe village derive around 20% of household income outside village boundaries. Future studies are therefore needed examining gradients which extend to greater extremes of deforestation, in order to explore whether household incomes become depleted at very low levels of forest or woodland resource availability. Extending the gradient into more completely deforested landscapes would also allow examination of how households respond when they cannot compensate for lack of resources in their own area by displacing resource collection to adjacent communities.

It is also important to critically assess our methods of quantifying provisioning ecosystem service availability. In order to understand variation in provisioning services we drew upon multiple data sets, including woodland surveys, focus groups and transect walks. Each of these data sets in isolation would provide only an incomplete picture: our woodland surveys quantified availability only of woody species, when herbaceous species are also locally important particularly as medicinal plants (Maroyi, 2011), and there are well documented issues in relying upon recall data alone to assess ecosystem service change. The triangulation of these

multiple data sets gives us confidence that our assessments of variation in provisioning service availability are robust; however, studies which extend to surveying the herbaceous species important to rural livelihoods and which quantify changes in the abundance of livelihood relevant species over time would be invaluable contributions to the literature.

5. Conclusions

Despite the caveats outlined above, we believe that the novel combination of methodological approaches in this study allows us to make a unique contribution to understanding of forest-livelihood interactions. This study demonstrates that while high biomass woodlands may make important contributions to rural livelihoods, they are by no means the only land cover important in the provision of environmental income, and the ability of households to continue deriving resources from ecologically degraded systems contributes to the observed resilience of household organic environmental incomes in the face of woodland loss. Any future landscape management interventions, such as reforestation programmes which may alter rights to 'degraded' woodlands (as per McElwee, 2009), will thus need to take into account the full complexity of land covers and collected resources within mosaic landscapes. We hope that this study will now form a valuable basis for further work quantifying both provisioning service availability and environmental income in complex forest-agriculture matrices at larger spatial and temporal scales, thereby enabling a much stronger understanding of how landscape structure and forest change interact with rural livelihood strategies.

Declaration of Competing Interest

The authors listed on this paper declare that they have no conflicts of interest which could have influenced the outcomes of this research.

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Appendix A. Supplementary data

Supplementary data to this article can be found online at <https://doi.org/10.1016/j.worlddev.2019.05.008>.

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