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Social as much as environmental: the drivers of tree biomass in smallholder forest landscape restoration programmes

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Abstract

A major challenge for forest landscape restoration initiatives is the lack of quantitative evidence on how social factors drive environmental outcomes. Here we conduct an interdisciplinary quantitative analysis of the environmental and social drivers of tree biomass accumulation across 639 smallholder farms restoring native tree species in Mexico, Uganda and Mozambique. We use environmental and social data to assess the relative effects of key hypothesised drivers on aboveground biomass accumulation at the farm-level over ten years. We supplement this with a qualitative analysis of perspectives from local farmers and agroforestry technicians on the potential causal mechanisms of the observed social effects. We find that the material wellbeing of farmers (e.g. assets) and access to agroforestry knowledge explain as much variation in biomass as water availability. Local perspectives suggest that this is caused by the higher adaptive capacity of some farmers and their associated ability to respond to social-ecological shocks and stresses. Additionally, the variation in biomass between farms increased over time. Local perspectives suggested that this was caused by emergent exogenous and stochastic influences which cannot be reliably predicted in technical analyses and guidance. To deal with this persistent uncertainty, local perspectives emphasised the need for flexible and adaptive processes at the farm- and village-levels. The consistency of these findings across three countries suggests these findings are relevant to similar forest restoration interventions. Our findings provide novel quantitative evidence of a social-ecological pathway where the adaptive capacity of local land users can improve ecological processes. Our findings emphasize the need for forest restoration programmes to prioritise investment in the capabilities of local land users, and to ensure that rules support, rather than hinder, adaptive management.

1. Introduction

Forest landscape restoration (FLR) initiatives are at the forefront of efforts to reverse environmental degradation in terrestrial ecosystems (Chazdon *et al* 2017). The success of FLR initiatives, however, has so far has been mixed (Aronson and Alexander 2013; Mansourian *et al* 2017).

A major challenge for restoration and other land management schemes is the difficulty of predicting, controlling and managing the outcomes of interventions in what are often highly complex and variable social-ecological systems (Messier *et al* 2015). There is ongoing debate on the drivers of FLR outcomes, with different perspectives giving varying levels of emphasis to environmental and social factors. Some emphasise biophysical aspects and the need to build and support the integrity of ecological communities—there may be social benefits, but objectives can be primarily ecological, knowledge is technical, and minimising human intervention is seen as key (J. C. Suding *et al* 2015, Brudvig *et al* 2017, Aronson *et al* 2018, Higgs *et al* 2018, Temperton *et al* 2019). Others emphasise the importance of

institutional and social contexts that support good governance and adaptive management for sustainable and socially beneficial restoration (Van Oosten 2013b, Mansourian 2016). This divergence of perspectives on the drivers of environmental outcomes also extends to the related fields of conservation and payments for ecosystem services (Soule 2013, Pascual *et al* 2014, Naeem *et al* 2015, Ezzine-de-blas *et al* 2016), and to the fields of land system science where existing models and approaches continue to struggle to integrate local-level social factors and context (Stephanson and Mascia 2014, Iwamura *et al* 2018). Effective interdisciplinary approaches to FLR and similar interventions remain rare (Huber-Stearns *et al* 2017, Mansourian *et al* 2017).

One of the key gaps in interdisciplinary FLR remains the quantification of how local (e.g. household-level) social factors drive biophysical outcomes, and clear knowledge on their causality (Wortley *et al* 2013, Chazdon *et al* 2017). While the field of restoration ecology has generated a wealth of quantitative empirical research on how environmental aspects drive outcomes (Perring *et al* 2015), due to the difficulty of measuring social phenomena, ex-post quantitative field studies testing the effects of social drivers have remained rare (Geist and Galatowitsch 1999, Miller and Hobbs 2007, Le *et al* 2012, Kibler *et al* 2018, Sapkota *et al* 2018).

In the land systems, forest transition and FLR literature, existing ex-post field studies that do cover the social drivers of biophysical outcomes have mainly focused on showing how socio-economic factors influence land users to join a scheme (e.g. Yin *et al* 2010, Mullan and Kontoleon 2012, Baynes *et al* 2017). While useful for targeting initial tree planting, there remains a dearth of field studies quantitatively assessing how social drivers effect biophysical outcomes (e.g. tree growth) at the local level. The few field studies that do assess biophysical outcomes have mainly focused on broad assessments of project-level factors such as institutional design and economic incentives, and have found that social drivers are secondary to environmental drivers (Yackulic *et al* 2011, Le *et al* 2014). However, such project-level assessment likely miss the great social diversity at sub-project (e.g. household) levels which likely has great effect on land management and tree care (Tittonell *et al* 2005, Nahuelhual *et al* 2018, Pritchard *et al* 2018).

A consequence of the lack of fine-grained social analyses, is that models and guidance for predicting and managing FLR outcomes are often focused on technical, largely environmental, factors (Wortley *et al* 2013). On the other hand, in implementation, land management schemes are challenged to contend with a much broader array of both social and environmental factors (Van Oosten 2013). Generating quantitative evidence on the relative importance and causal mechanisms of social factors remains a

research frontier for FLR and other land management interventions (Chazdon *et al* 2017).

Here we begin to address this gap through an novel ex-post, field-based interdisciplinary quantitative analysis of environmental and social drivers of tree biomass accumulation across 639 smallholder agroforestry farms restoring native tree species in projects in Mexico, Uganda and Mozambique. To our knowledge this is the first such quantitative analysis of its kind. Additionally, as we will elaborate, the consistency of our results across three countries strengthens the generalisability of our findings to similar land management interventions.

Agroforestry with native species is increasingly advocated as a key method of FLR, where farmers can increase native tree cover while maintaining crop production in agricultural landscapes (Erdmann 2005, Schroth *et al* 2011, Robiglio and Reyes 2016). Smallholders are estimated to manage approximately 75% of the world's agricultural land (Lowder *et al* 2016), and to make up most of the world's poor (Morton 2007). Thus, many FLR initiatives, and particularly those in developing countries, will engage smallholders—and native-species agroforestry offers a key way to do this.

We focus on five key environmental and social factors theorised (by both experts and local land users) to drive biomass outcomes in such interventions: water availability; soil quality; existing tree cover at time of planting; household wealth and living standards (henceforth 'material wellbeing'; White 2010); and household access to agroforestry knowledge. The environmental variables cover the key ecological considerations in designing agroforestry systems: sufficient water and soil nutrients are fundamental for tree growth, while tree cover at the time of planting serves as a proxy for inter-plant competition (Ashton and Montagnini 1999, Corona-Núñez *et al* 2018).

For social drivers, dimensions of household material wellbeing have been shown to be key factors in determining smallholder land management and resource use—people with different levels of deprivation have different capacities to manage land, and rely on different resources (Tittonell *et al* 2005, Nahuelhual *et al* 2018, Pritchard *et al* 2018). For access to agroforestry knowledge, both vertical (expert to farmer) and horizontal (farmer to farmer) extension services (Altieri and Toledo 2011) have been associated with the successful uptake of new land management techniques amongst smallholders (Clark *et al* 2011, Baird *et al* 2016).

More broadly, access to assets and knowledge are theorised to be central to the adaptive capacity, and associated resilience, of actors in natural resource management—a key factor underpinning the achievement of land management objectives despite emergent shocks and stressors (Thiault *et al* 2019). For FLR, social factors, extension services and
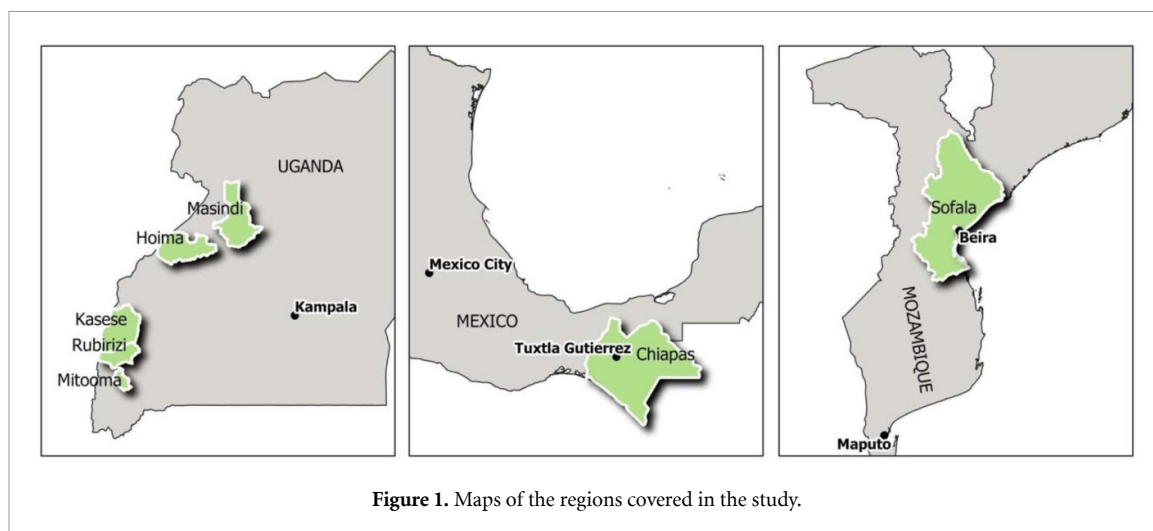
Figure 1. Maps of the regions covered in the study.

associated adaptive capacity are postulated to be key enabling factors for successful outcomes (Yin *et al* 2013, Chazdon *et al* 2017).

Our research questions are: which of the hypothesised environmental and social drivers have had the greatest effect on the AGB of trees established on agroforestry restoration farms? What are the causal mechanisms of the social effects? What are the implications for smallholder agroforestry, and other, FLR projects?

2. Methods

2.1. Study design

We use tree inventories, social surveys, spatiotemporal biophysical datasets, biomass modelling and mixed effects models to assess the relative effects of a set of hypothesised environmental and social drivers on the accumulation of aboveground biomass (AGB) at the farm-level across all three projects. We focus on AGB as a key metric for understanding changes in forest landscapes (Goetz *et al* 2015), acknowledging that the benefits of trees in these landscapes go far beyond biomass. We identified the hypothesised drivers with reference to both the literature and interviews with local farmers and agroforestry technicians (details below). We also used these interviews to supplement the quantitative analysis with local perspectives on the potential causal mechanisms of the observed social effects.

2.2. Study areas

Our study sites cover farms participating in three smallholder agroforestry schemes: Scolel'te in Chiapas State in southern Mexico; Trees for Global Benefits in the districts of Rubrizi, Mitooma, Kasese, Hoima and Masindi in western Uganda; and the Sofala Community Carbon Programme in Sofala Province in central Mozambique (figure 1). The farms in Mexico occur across a 240 km section of

the highlands in Chiapas, along an ecological gradient from montane tropical rainforests to subtropical pine-oak rainforests (De Jong *et al* 1995, p 99). Farmers are from a diverse range of villages, spanning five culturally distinct *Maya* linguistic groups, and *mestizo* farmers of mixed descent (Ruiz-De-Oña-Plaza *et al* 2011). In Uganda, sites occur along a 330 km section of the Albertine Rift characterised by crater lakes and tropical high forests. Farmers are members of a range of different *Bantu* linguistic groups (ECOTRUST 2018). In Mozambique, sites are spread across a 30 km area of tropical open miombo woodland (sometimes classified as savannah) bordering the Gorongosa National Park (Ryan *et al* 2011, Woollen *et al* 2012). Farmers generally share *Sena* as their local language and are comprised of both long term residents and refugees who have settled in the 1990s following the Mozambican civil war (Hegde *et al* 2015).

Each project implemented its own types of agroforestry with different species and management protocols, designed for different existing land uses and bioclimatic zones (table 1). The different existing land uses and species likely imply different natural growth rates, and different levels of tree management and care. To enable an analysis across agroforestry types and bioclimatic zones, we use a relative measure of biomass accumulation which controls for different land uses, species and management (see Methods). Each village in the project relied on its own nursery for tree saplings. Assuming sapling quality varies with nursery, to control for variation in sapling quality we nested our analysis at the village level.

While socio-ecologically diverse, all regions share similar levels of variance on the key variables in our analysis (table 2, in bold). Additionally, all can be categorised as remote areas dominated by subsistence agriculture and/or livestock systems, with high levels of poverty by global and national standards (OPHI 2015, 2018a, 2018b). All three schemes are funded by a mix of donor funds and carbon credits generated

Table 1. Summary of agroforestry types and land uses.

Agroforestry type	Description
Live fence, Mexico	Planting of *Cedrela odorata* around the edge of existing arable fields or areas of pasture. Initial minimum tree spacing of 3 m (a stocking density of 133 stems around a one ha. field).
Intercropping, Mexico	Establishment of forestry plantations of *C. odorata* and *Swietenia macrophylla* alongside existing annual agricultural crops. Initial minimum stocking density of 333 stems per ha.
Coffee, Mexico	Enrichment planting of *C. odorata* to provide shade in coffee plantations. Initial minimum stocking density of 180 stems per ha.
Improved fallow, Mexico	Planting of *Pinus oocarpa* and *Quercus spp.* on long term fallows or areas unsuited for agriculture. Initial minimum stocking density of 475 stems per ha.
Live fence, Mozambique	Planting of native hard wood tree species (typical of miombo woodland) around the edge of existing arable fields. Initial minimum tree spacing of 4 m (a stocking density of 84 stems around a one ha. field).
Intercropping, Uganda	Establishment of forestry plantations of *Maesopsis eminii* alongside existing annual agricultural crops. Initial minimum stocking density of 333 stems per ha.

under the Plan Vivo Carbon Certification system (Plan Vivo 2013). They thus have similar organisational processes and land management objectives, where a local organisation employs local technicians to help farmers to restore native tree species, and to monitor tree growth for 10 years after planting. These project processes are integrated with existing village institutions to varying degrees.

2.3. Sampling

We analysed 639 randomly-selected households and their associated agroforestry farms (259 in Mexico, 321 in Uganda and 59 in Mozambique). In Mexico and Mozambique, we excluded farms for which we had insufficient social variables. Assessments of missing values showed no structure to the missingness, implying values were missing at random—and thus that our overall sample can continue to be considered random (Kowarik and Templ 2016). Our sampling frame covers populations of farmers who opted to participate in FLR in three different countries. We therefore interpret our results as case studies having relevance to similar interventions (Yin 2014).

2.4. Data: relative aboveground biomass

To generate farm-level estimates of AGB per hectare, we used farm-level tree inventories, the pantropical allometric models provided by Chave *et al* (2009), (2014); and the BIOMASS package in R (Rejou-Mechain *et al* 2018). Tree inventories were census-style surveys, measuring all planted trees on the farms and recording species, tree diameter-at-breast-height (DBH; approx. 1.3 m), tree height, wood density and plot location. Height was recorded for all trees (including saplings), while DBH was measured for all trees with DBH $\geq$ 5 cm. The BIOMASS packages in R package accounts for variation in allometry by bioclimatic zone based on the expected location of the plot. We used Monte Carlo simulation to generate 95% credibility intervals (CI) of AGB on each farm.

Each project implemented different styles of agroforestry (table 1), with different tree communities for different bioclimatic zones, and so different expected rates of biomass accumulation. To enable comparisons of performance between agroforestry styles and bioclimatic zones, and plots of different ages we calculated a measure of relative aboveground biomass (RAGB). First, we used chronosequences (Walker *et al* 2010) and least square log-linear regressions (Paine *et al* 2012) to find the expected 'average' AGB per hectare for a particular year (up to 10 years since planting) for a given agroforestry style. We then extracted for each farm the adjusted standardised pearson residuals (i.e. the deviation of the farm AGB from the expected AGB, in standard error units; similar to a z-score) as an indicator of relative performance (Maschinski *et al* 1997, Kastenholz and Rogrigues 2007, Sorice *et al* 2014). We used the conservative RAGB value for each farm (the lower 95% CI RAGB for farms with mean RAGB > 0, and the upper 95% RAGB for farms with mean RAGB < 0, where RAGB = 0 indicates average performance).

2.5. Data: environmental explanatory variables

For water availability, we modelled the mean annual climatic water deficit (CWD; potential evapotranspiration minus actual evapotranspiration) since planting on each farm (for a similar approach see Poorter *et al* 2016) using farm location data, global spatiotemporal records of temperature and rainfall since tree planting (data from Willmott and Matsuura 2014; digital-elevation-model assisted interpolation from weather station records to 0.5 degree resolution), digital elevation models (INEGI 2018, USGS 2006; 30 m resolution) and the CWD R function from Redmond (2015). For soil quality, we used estimates of cation exchange capacity (CEC) from the ISRIC

Table 2. Descriptive statistics of variables. Variables in bold are included in the main model.

Variable	n	Mexico Mean ± SD (% for binary)	n	Mozambique Mean ± SD (% for binary)	n	Uganda Mean ± SD (% for binary)
Travel time to city (mins)	259	154.45 ± 84.18	59	225.42 ± 16.75	321	71.01 ± 23.68
Amount land (ha)	259	9.38 ± 6.74	59	1.51 ± 1.45	321	10.76 ± 14.67
Literacy	259	93%	59	44%	321	74%
Valuable assets (2nd model only)	259	52%	59	12%	83	29%
Above primary schooling 2nd model only)	259	53%	59	17%	60	25%
Employment contract (2nd model only)	106	8%	59	15%	85	11%
Formal land tenure	259	80%	59	51%	321	24%
People in household	259	4.27 ± 1.4	59	6.22 ± 1.92	321	8.71 ± 0.88
Wellbeing index (main model: simpler, full sample)	**259**	**3.93 ± 1.91**	**59**	**2.29 ± 0.89**	**321**	**1.99 ± 1.01**
Wellbeing index (2nd model only: broader, partial sample)	106	5.06 ± 2.13	59	2.73 ± 1.16	60	1.68 ± 1.13
Village AF experience (years)	259	4.61 ± 2.8	59	2.54 ± 2.28	321	2.5 ± 2.3
Technician in village	259	85%	59	36%	321	70%
Extension services index	**259**	**1.27 ± 0.47**	**59**	**0.59 ± 0.56**	**321**	**0.93 ± 0.55**
Tree cover at planting (%/ha)	**259**	**42.59 ± 13.06**	**59**	**10.04 ± 3.18**	**321**	**7.87 ± 2.36**
Cation exchange capacity (cmol +/kg)	**259**	**25.92 ± 3.54**	**59**	**9.38 ± 0.87**	**321**	**15.79 ± 3.49**
Mean climatic water deficit (mm/yr)	**259**	**−296.35 ± 139.11**	**59**	**−399.15 ± 119.75**	**321**	**−294.7 ± 128.5**
Initial planting density (stems/ha)	**259**	**426.85 ± 242.68**	**59**	**75 ± 6.27**	**321**	**365.09 ± 24.21**
Farm size (ha)	**259**	**1.01 ± 0.43**	**59**	**1.1 ± 0.94**	**321**	**1.67 ± 1.31**
Relative above-ground biomass	**259**	**0.01 ± 0.74**	**59**	**0 ± 0.57**	**321**	**0.01 ± 0.79**

SoilGrids global spatial datasets (Hengl *et al* 2017; from soil field measurements extrapolated using 158 remote-sensing-based soil covariates at 250 m resolution). For existing tree cover, we used farm locations and assessments of tree cover from spectral Landsat and MODIS remote sensing data (Sexton *et al* 2013; 30 m resolution) to estimate the proportion of tree cover on the plot in the year of planting. We also included the initial stocking density of tree on each plot as a supplementary measure of competition, and the size of the farm to check for bias from farm size (e.g. the overestimation of biomass on smaller farms).

Data on CWD, CEC and initial tree cover are at a coarser resolution than all other variables, which all operate at the farm-level or similar scales. The spatial mismatch between CWD and CEC and our

outcome measurements increases the likelihood of random error in the modelling, which would weaken their effects in the regression analysis. Nonetheless, we include these variables to assess whether broad variation in these soil and climate variables have an overwhelmingly large effect on biomass accumulation that renders social factors obsolete.

2.6. Data: social explanatory variables

For material wellbeing, we constructed an index of multi-dimensional material wellbeing using similar indicators and the same 'counting' approach as the widely-used global multidimensional poverty indicator (MPI; see Alkire and Jahan 2018). Data were sourced from household surveys conducted with the randomly selected farmers in each country. All surveys were conducted face-to-face with the person responsible for managing the farm (i.e. usually the farm owner). Interviews were conducted with the help of a local translator (see S1 in the supplementary material for further details) (available online at stacks.iop.org/ERL/15/104008/mmedia). We followed a similar approach to construct an index of access to extension services based indicators identified from local consultations and the existing literature (Krishna 2004, Birner *et al* 2009, Altieri and Toledo 2011). All quantitative variables are summarised at table 2.

2.7. Data: local perspectives on causality

To better frame our hypotheses, and to understand how social drivers operate, we conducted semi-structured interviews with 39 farmers and 23 technicians during field visits to Mexico, Uganda and Mozambique (29 in Mexico, 13 in Uganda and 20 in Mozambique). We used a purposive sample to speak to farmers with varying levels of AGB performance and the main technicians associated with those farms. We conducted these interviews as broad, semi-structured conversations about the respondent's experience throughout the project, including open questions on why some farmers have bigger or different trees compared to others. Interviews were conducted with prior informed consent and anonymity was maintained throughout. We documented interviews in notes and audio recordings, sometimes with the assistance of translators fluent in the local languages.

2.8. Analysis

For the quantitative analysis, we used linear mixed models with REML estimation, and village and country as a random effect (minimum of 12 households per village). Diagnostics indicated a suitable fit with normally distributed residuals with homogenous variance and no significant collinearity among independent variables (Zuur *et al* 2007). We also subsequently conducted a likelihood ratio test to check the significance of the random effect of village (Kuznetsova *et al* 2017). Given the varying resolution of the variables in our analyses, we used variograms to assess the spatial dependence of all independent variables and the dependent variable (RAGB), and global tests of Moran's I and correlograms to assess spatial autocorrelation in the residuals of the main model. We also plotted model residuals against farm size to check for bias in biomass estimates from large trees on small farm sizes. All analyses were performed in R, version 3.5.1 (R Core Team 2019), and the model code and diagnostics are in the supplementary material, section S3. For the qualitative analysis, we used a thematic analysis (Ritchie *et al* 2013) to frame the hypotheses around material wellbeing and agroforestry knowledge and, following the quantitative analysis, to examine in more depth the possible causal mechanisms behind the observed social effects. We include illustrative (anonymised) quotes from respondents in the results.

3. Results

Across our sites, farm-level AGB varied greatly, and this variation increased over time (figure 2).

Box 1. Local perspectives on social-ecological diversity

Every farm is different. The soil changes from one farm to the other. Some are closer to the (existing rainforest) so they get more vines and shade. People also want to do different things on their farms.

Farmer, Mexico

People are not the same, so having one (agroforestry) plan does not work. You need several options with some flexibility. Some people like different trees because of the fruit or medicines. Also some trees grow better in some places but we do not really understand why. Even the (forest ecologists) do not know.

Agroforestry technician, Uganda

Perspectives from farmers and local technicians suggested that this reflects the great and inherent social-ecological diversity amongst smallholdings, even across small areas (Box 1).

Local actors also suggested that following the establishment (tree planting) phase, land managers will lose control over outcomes as emergent social-ecological factors outside of their influence come to bear (Box 2).
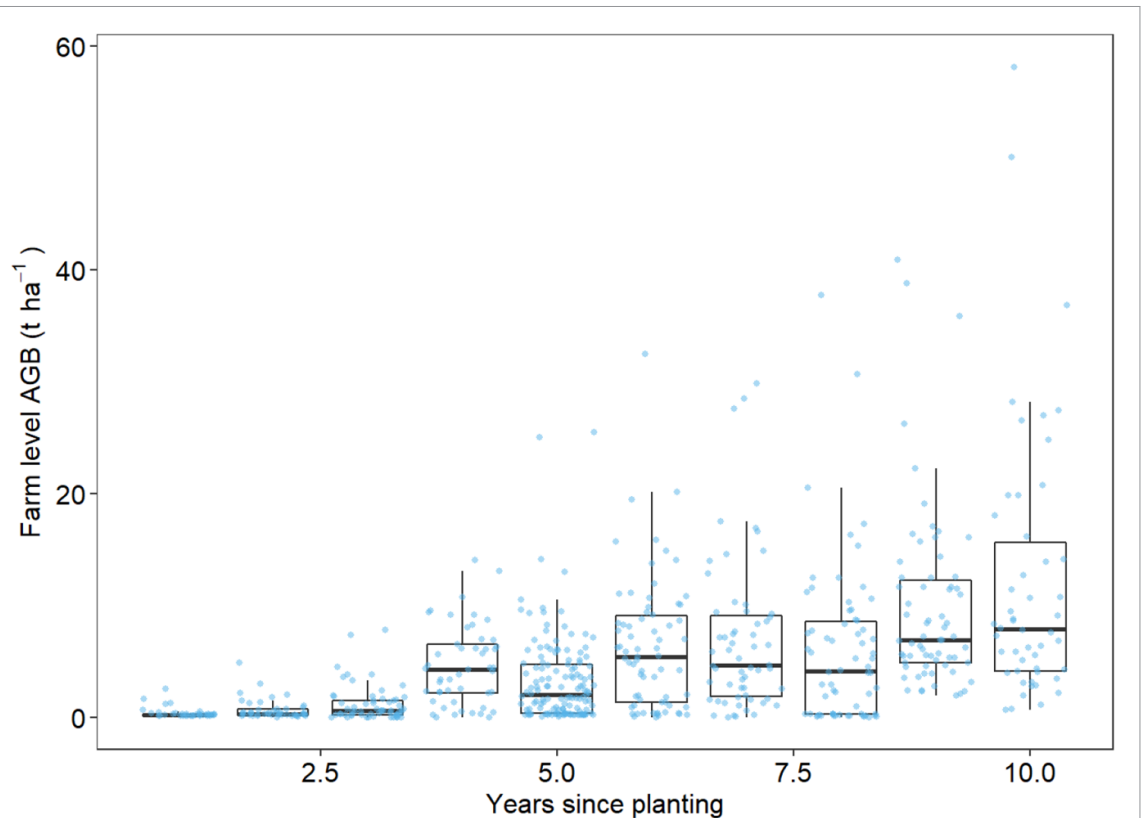
Figure 2. Boxplots showing variation in aboveground biomass between farms of different ages. The boxplots show quantiles, while the points are individual farms (horizontally jittered to the width of the boxplot). Tree stocking densities are a main determinant of AGB per ha, and target stocking densities varied between the different agroforestry styles included in the study. Here we show farm-level AGB for all land uses, normalised to a stocking density of 100 stems per ha.

Box 2. Local perspectives on a loss of control over emergent social-ecological factors

There have been big social and environmental changes since the beginning of the project. In some places there were floods, and in other years there were small fires. Other years it was ok. Also there are now more people and less land. (The project processes) had to change but you cannot control everything.

Agroforestry technician, Mexico

It was easy (to grow trees) at first, but then some (farms) do better than others. We had a dry year, so people that had just then planted now have smaller trees. Some people did a better job at watering (the saplings), but even then that did not always work.
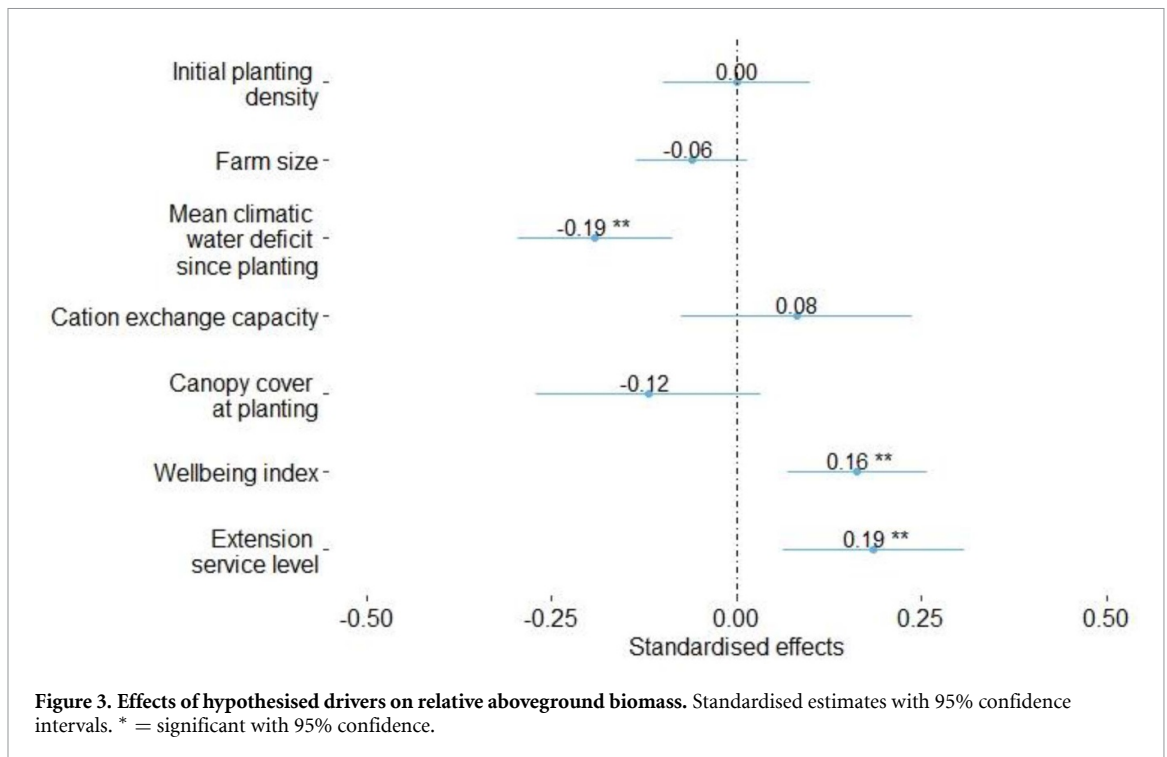
Farmer, Mozambique

In the regression analysis, the social factors of household material wellbeing and access to extension services each explained similar amounts of variation in RAGB to that explained by climatic water deficit

(figure 3). Cation exchange capacity, tree cover and initial planting density had no significant effects. The relative homogeneity of residuals across countries (supplementary material, section S3a.i), and supplementary individual regressions for the limited sample sizes in each country (supplementary material, Section S3b), indicate that these results are robust across our sites. Additionally, farm size had no apparent influence on the model residuals (S3a.ii), indicating that the results are robust to the influence of large trees on small farms.

Our results also appear robust to spatial autocorrelation (supplementary material, section S3d). While variograms indicate strong spatial dependence of some of our environmental independent variables (CWD, CEC and initial tree cover), all other independent variables and our dependent variables appear strongly spatially independent. Crucially, correlograms of Moran's I of model residuals found no significant spatial dependence at different spatial lags in Uganda and Mozambique, and only a very weak dependence at very large spatial scales in Mexico (Moran's I = 0.05, p < 0.01, at distance class midpoint of 1.33 decimal degrees; 148 km at the equator).

Broadly, these results suggest that social factors have a measurable impact on biomass accumulation. Given that variability in AGB increases over time and that we only model growth in the first ten

Figure 3. Effects of hypothesised drivers on relative aboveground biomass. Standardised estimates with 95% confidence intervals. * = significant with 95% confidence.

years since planting, effects are likely to be greater by the time trees reach maturity (25 to 40 years). Our conclusions on the relative influence of the environmental factors of CWD, CEC and initial tree cover are limited by the coarser resolution these variables. However, we view that the lack of significantly larger effects of these environmental variables relative to social variables does emphasise that both are integral to biomass accumulation in FLR schemes.

The inclusion of village as a random effect significantly improved the model fit ($X^2 = 46.77$, df = 1, N = 639, p < 0.01), indicating that farms associated with the same village performed similarly. Conversely, however, there was low spatial auto-correlation of RAGB in Mexico (Moran's I = 0.23, p < 0.01) and Uganda (Moran's I = 0.14, p = 0.02) (Mozambique had an insufficient sample for a robust assessment). These results combine to indicate that there are additional drivers operating at the village level and that they are not strongly spatial.

These statistical associations correspond with the consistent perspective amongst farmers and technicians that farmers with greater individual capabilities, and more supportive village institutions, were better able to innovate and adapt their land management in response to changing social and environmental conditions. Essentially, farmers with sufficient capabilities appear more able to overcome environmental barriers to tree growth by having more time, labor and knowledge to allocate to the care of their trees (Box 3)

Box 3. Local perspectives linking social factors, adaptive capacity and tree growth

It is easier for richer people, or people with a bigger group to help, because they have more labour … and money is also important. When things happen, you can use the money to deal with it.

Farmer, Mexico

It was difficult because it was hard to do something new. Some of the trees did not work because of the drought, then my husband got sick and it was difficult to fix things

Farmer, Mozambique

It was always harder when there is no one else doing agroforestry in the village. Farmers need to learn what works and this is always easier in a group, or when someone has done it already.

Agroforestry technicians, Uganda

I lived next door to the house where the (agroforestry technicians) would stay. It helped to have them next door. They would always come and give advice which helped the trees.

Farmer, Mozambique

More broadly, while our modelling showed some significant effects, most of the variation in AGB remained unexplained, despite the fact that we had accounted for (to the best of our ability) the major drivers suggested by local stakeholders and the technical literature. While our use of a relative measure of biomass accumulation, and the nesting of our analysis at village level, controls for broad differences in species, sapling quality and land management, residual variation is likely explained by other ecological (e.g. disturbance; species interactions; microclimates) and social factors (e.g. the nuances of household participation in resource governance institutions; within-household interactions) not covered in our analysis. Combined with local perspectives on the inherent variability and dynamism of the social-ecological system (Box 1), this suggests that there are no simple explanations for variation in land management outcomes in our systems—drivers are likely diverse and very hard to measure and predict. In this context of continued uncertainty, local perspectives emphasised the importance of adaptive learning at the project, village and farm levels. As an agroforestry technician in Uganda told us: 'New things arrive in the project that you cannot anticipate. So we need to be flexible if we can, while still caring for the trees and forest. When changes come, we all change as one.'

4. Discussion

In this study, we find strong quantitative evidence that the material wellbeing and knowledge of farmers are key drivers of biomass accumulation in smallholder agroforestry FLR interventions. To the best of our knowledge, this phenomenon has not previously been demonstrated quantitatively using ex-post field data linking directly to biophysical outcomes. Additionally, the quantitative evidence suggests that these factors operate at both the village and household levels.

Local perspectives emphasised that the broad causal mechanism for these social effects was that farmers with more resources and knowledge, and better support from village institutions, were better able to adapt their land use to emergent social-ecological shocks and stresses. This reaffirms existing theories on the importance of individual adaptive capacity and adaptive cogovernance for land management programmes (Thiault *et al* 2019).

Our findings apply across sites in three countries. Given the need for FLR and other restoration programmes to engage rural smallholders in developing countries, we contend that our results are of relevance to the broader restoration field, and other land management interventions such as conservation and payments for ecosystem service schemes. Below we highlight two key contributions.

4.1. Social resilience and adaptive capacity drive restoration outcomes

A part of the restoration literature continues to view social factors and objectives as secondary (albeit admirable) considerations for restoration initiatives, relative to more important biophysical considerations (Aronson and Alexander 2013; Suding *et al* 2015, Higgs *et al* 2018, Temperton *et al* 2019). This view is also prominent in part of the associated conservation and payments for ecosystem services literatures, where social objectives are sometimes seen as aspirational but not integral (and sometimes as a distraction) to technical and biophysical factors (Soule 2013, Naeem *et al* 2015, Ezzine-de-blas *et al* 2016).

Our results provide robust empirical evidence demonstrating that the social situation of local resource users has a significant, tangible effect on biophysical restoration outcomes. This accords with existing literature on the importance of social factors supporting good governance (Mansourian 2016, Van Oosten 2013; Baynes *et al* 2017), and extends this to emphasise the importance of supporting the adaptive capacity of individual participants. It also contrasts with coarser (e.g. project-level) analyses which have found no effect from social factors on biomass accumulation in FLR project (Le *et al* 2014). By analysing at the household-level we have uncovered novel evidence on how social diversity drives biomass outcomes.

While improvements in ecological processes are often theorised to benefit humans (Díaz *et al* 2018, Chazdon and Brancalion 2019), here we have clear evidence of a reciprocal pathway: in certain contexts improvements to human capabilities can benefit ecological processes. Essentially, the effectiveness of a land management intervention may only be as good as the social-economic resilience and adaptive capacity of its local participants. Restoration, and related conservation and payments for ecosystem services projects, should thus put such factors on par with biophysical and other technical considerations.

One interpretation of this finding could be that restoration and similar programmes should avoid engaging poorer people with low capabilities. However, where interventions are aiming for a socially beneficial and landscape-level transformation, excluding more vulnerable people is likely not an option. On the social side, interventions would need to consider the social impacts of excluding already vulnerable and marginalised people from natural resource management programmes, and the related risk of elite capture (Persha and Andersson 2014). Excluding particular actors could also have knock on effects on community support for the project, and associated local perceptions of project legitimacy (Pascual *et al* 2014). Regarding landscape-level transformation, excluding particular actors could restrict interventions to site-level rather than landscape-level interventions, which would likely not achieve the changes that many hope for (Lamb *et al* 2005,

Chazdon *et al* 2016). It could also drive 'leakage' where conservation of one place in the landscape just moves degradation elsewhere (Bode *et al* 2015). Programmes seeking socially beneficial, landscape-level change will thus likely need to engage many actors, including vulnerable people. Allocating resources and designing institutions to supporting the adaptive capacity and capabilities of local resource users will be key. This will be particularly important for engaging smallholders, who are often poorer and control much of the world's land (Morton 2007, Lowder *et al* 2016).

4.2. Accepting uncertainty and supporting adaptive management

A second key finding of our study is that great variability in land management outcomes may be the norm rather than the exception in smallholder FLR and similar projects, even amongst sites in similar areas with similar land use objectives. Further, this variability likely increases over time. Local perspectives suggest that, rather than technical staff and FLR administrators progressively refining their knowledge and management of the system to reduce variability in outcomes, such actors may in fact begin to lose influence over land management outcomes after the initial establishment of the system. After this, exogenous and stochastic influences may come to dominate, and early differences in the quality of tree planting are exacerbated, pushing the system beyond the predictive and managerial control of land analysts and users.

Alongside our findings about local adaptive capacity, this emphasises the need to moderate expectations of being able to accurately design and predict interventions and outcomes (Brudvig *et al* 2017). Instead our evidence supports calls to invest in flexible rules and institutions that support rather than hinder adaptive management in restoration and related initiatives (Murray and Marmorek 2003, Mansourian *et al* 2017). Adaptive management is increasingly argued to be key for dealing with uncertainty and complexity in social-ecological systems (Schultz *et al* 2015), and our quantitative and qualitative findings support such an approach. This speaks to an ongoing tension in the restoration and conservation literature between those who wish to standardise 'best practice' approaches, and those who wish to maintain flexibility (Aronson *et al* 2018, Higgs *et al* 2018, Wunder *et al* 2018). Our findings support adaptive management as one of the core principles of FLR (Besseau *et al* 2018). We contend that all initial designs and predictions of restoration and other land management projects are likely to turn out to be at least a little inaccurate in practice—investing in adaptive project processes to adjust and correct interventions over time will therefore be key.

5. Conclusion

Our work offers novel evidence on the importance of social factors in driving outcomes in FLR and similar initiatives. We have shown across several hundred farms in three countries that the capability and knowledge of land users can drive outcomes alongside environmental factors—and that this is likely tied to the capacity of land users to respond and adapt to social-ecological shocks and stresses. While there are no doubt many other drivers of outcomes in our sites, and while the magnitude of the effects will likely vary across contexts, we argue that the consistency of our findings across three sites strengthens their relevance for other sites and programmes.

Broadly, we contend that restoration initiatives and similar land management programmes must build and maintain the adaptive capacity of smallholders and other local actors through both material and institutional support. Additionally, project designs, funding and rules must be flexible enough to support adaptive management in the context of continued uncertainty. Overall, we suggest that the field of 'restoration ecology' must become 'adaptive restoration social-ecology' if it is to succeed.

Data availability

The data of the study are available upon request to the authors.

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