Assessing the welfare impacts of forest ecosystem service management policies and their distributional rules

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Abstract

Community based management (CBM) is widely advocated as an effective method for governing and managing ecosystem services (ES). However, the distributional rules and maximum harvesting levels are likely to affect both the effectiveness of CBMs in maintaining ES and the fairness and equity of access to these ES. This paper proposes a methodological approach for investigating normative trade-offs involved in CBM of forests, where forest conservation objectives need to be traded off against livelihoods objectives. The study uses remote sensing methods to quantify forest ES supply in Namizimu Forest Reserve in Malawi, and links this to demand for ES within the villages near the reserve. It then investigates how a plausible set of CBM rules can be developed to cap consumption of forest products to sustainable amount and quantifies, by using monetary valuation techniques, how these set of rules may affect the total well-being of local population.

Our results demonstrate that, due to the spatial mismatches between demand and supply, the distribution of provisioning ES to the population across the harvesting area is unequal in biophysical terms. The current available stock of forest products is sufficient to cover the current demand, however it is higher than the mean annual increment indicating that this level of consumption is ecologically unsustainable and will lead to forest degradation as shown under the business-as-usual scenario.

We then examined the impact of governance and how CBM rules to allocate forest ES to different social groups (poor and rich) under a co-management regime will affect total societal welfare. We found that the distributional scenario that maximizes total societal welfare expressed in monetary terms across the whole harvesting area is the scenario that distributes 40% of biomass to the rich group while the remaining 60% is allocated to the poor group. However, this scenario maximizes Willingness To Pay (WTP) at total level but does not maximize WTP in each sub-area of forest but just for those that have a high availability for biomass. This indicates that the distributional rules that maximize total welfare at aggregate level may not maximize welfare at local level where constraints from biomass availability require to restrict further the distribution of forest products. When biomass availability is low, total societal welfare is maximized with distributional rules that distribute more trees to richer. Yet, a policymaker may choose a distributional rule that distribute more trees to the poor on normative grounds and forego the objective of maximizing total welfare. In such cases the WTP analysis outlined in this paper can support the policymaker in choosing the distributional rule that minimize trade-offs between efficiency, i.e., maximizing total welfare, and livelihoods objectives.

1. Introduction

Sustainable governance of natural ecosystems requires managing trade-offs between the conservation of ecological resources and the consumption of those resources to fulfil human population needs. Such trade-offs are especially relevant in low-income countries, where rural communities are strongly reliant on natural ecosystems for their basic needs and income (Angelsen et al., 2014; Fisher et al., 2014; Guerry et al., 2015) and the provision of ecosystem services play an important role in poverty alleviation (Daw et al., 2011, 2015; Ryan et al., 2016). In these contexts, the ecosystem governance mechanisms determine how these ecosystem services are allocated to different social groups by defining the rules and the rights for ecosystem users to access the resource and such mechanisms therefore have an impact on the distribution of these resources and its equality.

Attention to inequality and social injustice in ecosystem management and conservation is growing (Chaplin-Kramer et al., 2019; Martin et al., 2020). Different ecosystem governance arrangements lead to different outcomes for people and nature, as they adjust rights and responsibilities in ecosystem conservation, and seek to resolve resulting normative trade-offs in different ways (Mace et al., 2018; Sikor et al., 2014). Normative trade-offs between achieving ecological sustainability and improving or safeguarding the livelihoods of users of the natural ecosystem, remain understudied in the ecosystem services literature (Lautenbach et al., 2019; Schaafsma et al., 2021). Few ecosystem assessment studies present disaggregated analyses and evaluate distributive trade-offs among stakeholders to understand whether ecosystem governance projects that may seem to provide net positive gain overall also turn out to be positive for all actors involved (Kremen et al., 2000; Liu et al., 2017; Peh et al., 2014). Therefore, an environmental justice lens applied to the analysis of ecosystem service (ES) governance can reveal and evaluate distributional patterns arising from different governance mechanisms and provide support for decision-making where such normative trade-offs exists (McDermott et al., 2013; Schlosberg, 2013; Sikor et al., 2014).

This paper aims to examine the relevance of such normative trade-offs focusing on a case study on the management of a forest ecosystem in Malawi. In the study area, forest resources have an important role in supporting the livelihood strategies of rural communities, especially for the poorest village members, and can have an important equalising effect. Forest management policies in Malawi are transitioning toward community-based management (CBM) agreements and co-management for public forest reserves where local rural communities become responsible, in partnership with the forest department, for coordinating management and harvesting of forest resources. CBM is a widespread institutional mechanism for managing natural resources such as forests and fisheries in low-income countries (Agrawal et al., 2008; Sunderlin et al., 2008).[1] In CBM, local ecosystem users are involved in the definition of management rules for the natural ecosystem, thereby aiming to both improve ecological outcomes and facilitate access to all local users (Agrawal, 2001). The hypothesis underlying CBM is that local communities have the incentives and the knowledge to manage sustainably the natural ecosystem as well as establish management objectives and rules which accommodate all users' needs (Bowler et al., 2012; Ostrom et al., 1999). The

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[1] Here, we define CBM to encompass management forms with communities as the sole decision-makers, as well as co-management forms, in which rights and responsibilities divided between communities and other actors, such as national governments (e.g., co-management).

93   corollary from this is that CBM governance mechanisms are expected to solve conflicts arising from
94   normative trade-offs and generate equitable outcomes for all forest users.

95   In reality, whilst CBM has been found to improve ecological sustainability and reduced unsustainable
96   harvesting in multiple cases, results suggest it may be less effective in terms of achieving an
97   equitable distribution of benefits and improve livelihoods (Chomba et al., 2015; Dawson and
98   Coolsaet, 2015; Lund and Treue, 2008; Mutune and Lund, 2016; Persha and Andersson, 2014). Such
99   inequities are associated with low levels of active engagement of forest users, which can lead to a
100   limited development of equitable rules, where the interests of more marginalized groups are
101   excluded and elite capture can further exacerbate existing inequalities (Persha et al., 2011; Adhikari
102   et al., 2014; Chomba et al., 2015; Yadav et al., 2015). Moreover, harvesting limits, although
103   necessary to guarantee long-term ecosystem service supply, may put overly high opportunity costs
104   on local users (Green et al., 2018). As such, CBM may generate an unequal distribution of forest
105   resources and trade-offs in well-being across different social groups, increasing inequalities may also
106   decrease compliance and negatively affect nature conservation (Dawson et al., 2017).

107   This paper investigates the normative trade-offs between local welfare and forest conservation
108   objectives for CBM governance systems when the ecological constraints on ecosystem services
109   supply, the potential (mis-)match between supply and demand levels for these services, and the
110   distributional preferences of forest users are taken into account all together. It uses an
111   interdisciplinary approach, in which preference surveys among community members are combined
112   with a spatial analysis of forest stocks and flows, to better understand the implications of different
113   hypothetical co-management policies and their distributional rules for local welfare and actual
114   ecosystem services benefits.

2. Material and methods

116   This study employs a 3-step methodological approach to examine the welfare impacts of
117   distributional rules of hypothetical co-management policies in our case study - Namizimu Forest
118   Reserve in Malawi. We first give a broad outline of the methodological approach (section 2.1),
119   before briefly describing the case study (section 2.2), and then providing detailed methods for each
120   step (section 2.3).

2.1. Methodological approach

122   Identifying the actual contribution of nature to human well-being requires an understanding of the
123   potential demand of the human population for ES and how this demand is connected to the
124   potential supply of those services across space (Ala-Hulkko et al., 2019; Bagstad et al., 2013; Syrbe
125   and Walz, 2012; Vallecillo et al., 2019). The potential supply of ES depends on the structure and the
126   ecological processes of the natural ecosystem and is characterised by spatial variation (Barò et al.,
127   2016; Burkhard et al., 2012; Syrbe and Walz, 2012). Such potential supply becomes an actual ES if
128   there is a population that has demand for those services, can access it and benefit from it and thus
129   hold values for it (Potschin and Haines-Young, 2011). The ability of the population to benefit from
130   ecosystem services depends on the spatial connection with the areas that supply the forest
131   resources, i.e. the ecological endowment vis-à-vis the total demand of the population and their
132   distributional preferences (Fisher et al., 2014; Syrbe and Grunewald, 2017). Therefore, to assess the
133   welfare impacts of natural resource management on local welfare, we first need to quantify both the

134 potential supply and the potential demand as well as their spatial connection to quantify the actual
135 flow of ES, and then quantify the total value placed by people on the estimated actual flow of ES.
136 The three main methodological steps in this study are:

137 1) Step 1: evaluate the current level of forest degradation and quantify and map the potential
138 supply of ES, i.e., woody biomass available, and potential demand of ES, i.e., current and
139 expected consumption of woody products by the local population.
140 2) Step 2: model the spatial connection between potential demand and supply to quantify the
141 actual flow of ES that can be consumed.
142 3) Step 3: scenario analysis to assess the ecological consequences of current ES demand and
143 welfare implications of co-management policies across different distributional scenarios.

144 **Step 1 – Mapping and quantifying ES supply and demand**

145 The development of approaches for mapping and quantifying ES in semi-arid systems, such as
146 African woodlands, is still in its infancy compared to the more frequently analysed tropical and
147 temperate areas (Egoh et al., 2012; Wangai et al., 2016; Willemen et al., 2018). Most of the studies
148 on mapping ES have assessed the provision of regulating services such as carbon storage (Batjes,
149 2008; Egoh et al., 2011; Leh et al., 2013), water flow regulation (Egoh et al., 2008), soil accumulation
150 and retention (Egoh et al., 2011; Leh et al., 2013). Mapping and quantifying woody biomass provided
151 by forest ecosystems, when lacking primary data such as national forest inventories, relies on
152 indicators derived from remote sensing data combined with ground observations (DeFries et al.,
153 2007; Ryan et al., 2012). Therefore, to quantify and map the total biomass supplied by the forest
154 ecosystem analysed in this paper, we combine high resolution remote sensing images that measure
155 changes in vegetation cover with biomass estimates from ground observations. We use woody
156 biomass as a proxy to measure the provisioning ecosystem services of interest: fuelwood, poles, and
157 timber provision. To assess the current situation of our case study area we examined the current
158 ecological condition of Namizimu forest reserve by mapping the current level of degradation. To
159 quantify and map the future demand for biomass, i.e., potential demand, we use the current
160 average consumption of forest products of the local population. This consumption is quantified by
161 using primary data on household consumption of forest products collected through a household
162 survey administered to a sample of the population living in the surrounding of the forest ecosystem
163 under examination (Wolff et al., 2015).

164 **Step 2 – Linking supply and demand**

165 As forests (that supply ES) and village members (that demand ES) are not located in the same place,
166 we employed a spatial modelling technique to model their connection and quantify the actual flow
167 of ES. We used accessibility analysis to determine the potential link between the service providing
168 area, i.e. the forest, and the reference population (Ala-Hulkko et al., 2019; Syrbe and Grunewald,
169 2017). By simulating the spatial connectivity between supply and demand, we identified the actual
170 flow of ES and we evaluated where demand could exceed ES provision. This revealed where the
171 supply of ES would not be able to meet demand in the future and harvesting could become
172 ecologically unsustainable. By using a spatial approach, we also took into account the consequences
173 of spatial heterogeneity in the ecological structure of the forest as well as in population density and
174 desired amount of forest resources.

175 **Step 3 – Scenario analysis of co-management distributional rules**

176 The third step of our analysis focused on assessing the contribution to welfare of provisioning ES
177 under alternative distributive scenarios using Willingness-to-Pay (WTP) analysis. We used a Discrete
178 Choice Experiment (DCE) to elicit individual preferences for different level of biomass supply under a
179 CBM policy (Villamagna et al., 2013; Wolff et al., 2015). Based on welfare theory, DCEs are an
180 economic valuation method that can be used to assess respondents maximum WTP as a measure of
181 'strength of preference' for, or value of, a commodity, in monetary terms. We employed a novel DCE
182 designed to measure user's preferences for different distributions of biomass across different social
183 groups defined by their wealth status, i.e., better-off, and worse-off village members, and thus
184 assess the users' distributional preferences. Forests are common-pool resources characterized by a
185 conflict between the personal interest of forest users to maximise resource harvesting, and societal
186 interests in managing the resource sustainably to guarantee a long-term flow of benefits for all users
187 (Perman et al., 2003). As a consequence, individual preferences and behaviour regarding forest
188 management may be driven by ethical and moral considerations such as fairness concerns. This
189 raises the question of how members of forest-adjacent communities would prefer to see such
190 fairness considerations addressed in forest management, i.e., how they would want to distribute the
191 limited amount of biomass that can be harvested sustainably across current village members. Our
192 DCE asked respondents to make choices across different forest management scenarios, involving
193 different levels of private forest resources, different allocations of forest resources to others, and
194 different payment levels. The trade-offs between changes in forest resources and payment levels
195 results in individual WTP estimates. By aggregating the individual WTP for these options across the
196 full population, we could quantify in monetary terms the total welfare impacts associated with
197 different distributive scenarios. By using one metric, WTP, we were hence able to assess in monetary
198 terms the value of the trade-offs between social groups which arise from different institutional
199 arrangements (rules).

2.2. Case study: co-management in Namizimu Forest Reserve

201 Rural communities in Malawi strongly rely on forest ecosystem services. Forests are the primary
202 source of energy-related goods, i.e. firewood and charcoal; other important forest products include
203 construction materials, timber, wild food, and medicines (Campbell, 1996; Kamanga et al., 2009;
204 Lowore, 2003). Moreover, forests provide important indirect benefits such as flood protection and
205 soil fertility, regulation of the availability of water in streams during the dry season, and cultural
206 benefits. Forest resources are an important source of income for most rural households,
207 contributing about 10-20% of household income, and have an important equalizing effect (Angelsen
208 et al., 2014; Chilongo, 2014; Fisher, 2004). At the same time, forest cover in Malawi has been
209 declining at a fast rate; although the actual forest cover is not known with certainty, it has been
210 estimated that the annual deforestation rate ranges between 1% and 3.5% (Bekele, 2001; FAO,
211 2010; Zulu, 2010). As a mean to pursue effective forest management, the Malawian government has
212 introduced co-management of forest protected areas (GoM, 2001). This co-management policy
213 involves the sharing of rights and responsibilities about forest reserve between government and
214 local resource users. In the Malawi co-management policy, the local communities are involved in the
215 management of a delimited block of the forest reserve and through the election of a Village Forest
216 Committee (VFA), the local communities coordinate all harvesting activities and the distribution of
217 all forest products among local users.

218 In this study, we focused the spatial and scenario analysis on the area of Namizimu forest reserve
219 and surrounding villages. We chose Namizimu forest reserve as it is currently managed fully by the

220   forest department and co-management is not implemented yet, but it may be implemented in the
221   future. In this area, forest products are used almost exclusively for domestic consumption. Firewood
222   and charcoal are used as main energy source while poles and timber are used as construction
223   material. Briefly, the development of a co-management scheme requires identifying the interested
224   communities of local users, evaluate their needs and potential demand for forest products, carry on
225   an ecological assessment of the forest reserve areas to evaluate its potential supply and based on all
226   this information, develop a sustainable management plan. The communities of local users involved
227   in co-management schemes are usually selected between those located within a 5km buffer from
228   the forest reserve and are grouped at the administrative level of group village. Each of these user
229   group is then assigned a portion of forest reserve to manage, known as a "block" The management
230   partnership between the local community and the government becomes effective through a signed

231 binding agreement between the forest department and the residents of the communities involved
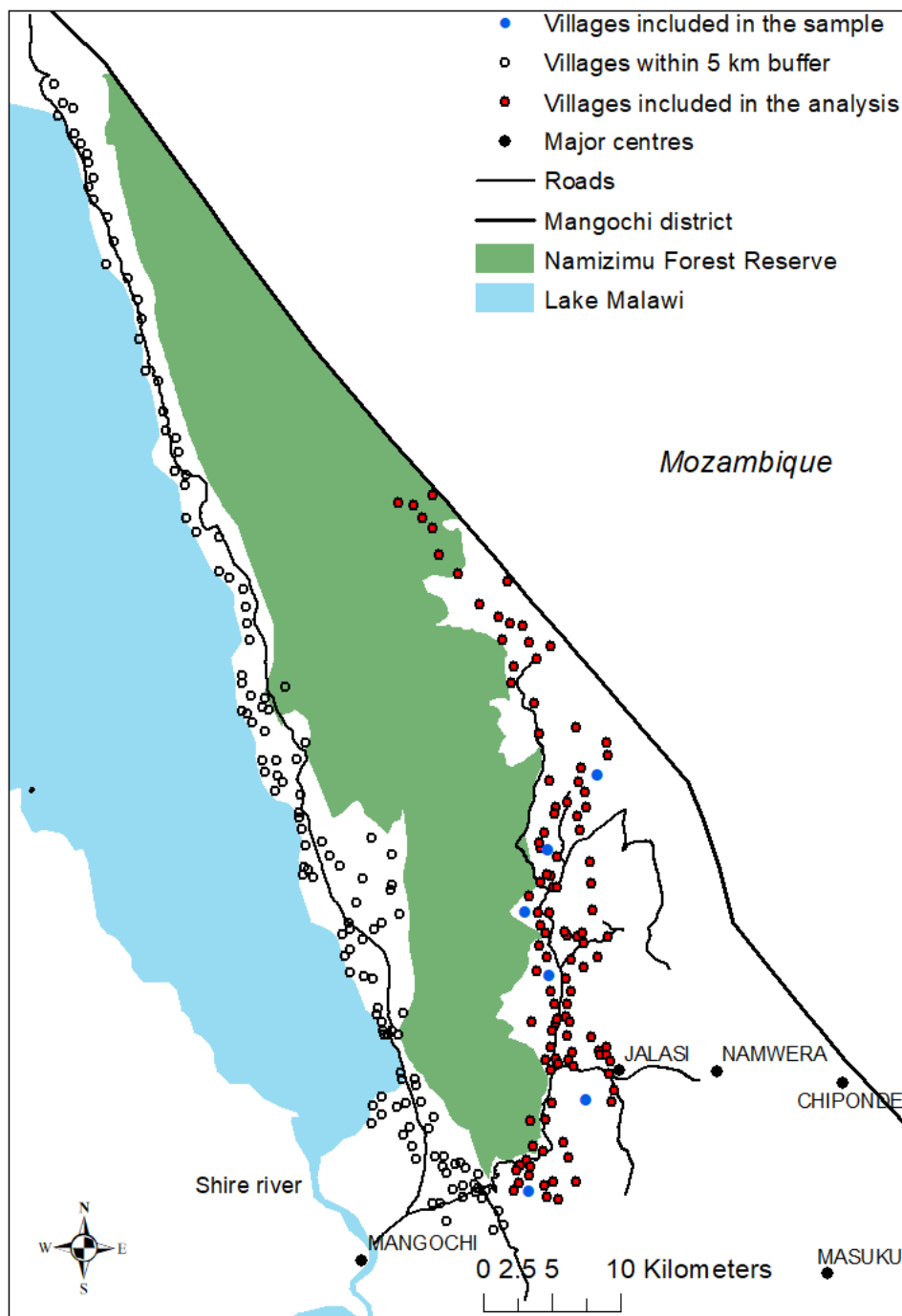232 which are represented by a committee (GoM, 2016, 1997).

233 Figure 1 - Case-study area. The map includes a visualization of Namizimu Forest Reserve boundaries
234 and all the villages surrounding the area that are located within a 5km buffer. The villages included
235 in the scenario analysis are in red while the villages where we collected primary data on
236 consumption of forest products through household survey are in blue. The location of the blue
237 villages is slightly shifted to guarantee compliance with research ethic requirements.

2.3. Methods and data

2.3.1. Step 1: spatially explicit quantification of supply and demand of biomass

Assessment of current forest degradation

In Malawi, the assessment of forest biomass stock in forest reserves which is currently publicly available is based on estimates which employ data from the two forestry inventories carried out in 1973 and in 1991 (FAO, 2010) and data on periodic field measurement in permanent plots are not available. Therefore, to provide a more updated assessment of the current ecological condition of the whole Namizimu forest reserve, we used a degradation index (Baumann et al., 2014) estimated using satellite images of the forest area. The degradation index employs a qualitative approach to examine changes in biomass stocks and providing an overall assessment of degradation patterns within the study area for the period 2013 to 2018. This qualitative approach are often used when accurate ground data information on the current forest stocks and, more importantly, how it changes over time are lacking (Mitchell et al., 2017).

The disturbance index is a spectral index calculated using a tasselled cap-based transformation, i.e. a remote sensing technique used to estimate vegetation condition from satellite images (Healey et al., 2005). The approach used by Neil (2018) allow to produce maps of the forest area where for each pixel the level of degradation is assigned through comparison to a chosen threshold that is completely disturbed (Healey et al., 2005). Specifically, this tasselled cap-based transformation reduces the remote sensing images obtained from Landsat satellite reflectance bands to three distinct tasselled components (Brightness, Wetness and Greenness) and evaluates how much these components differ compared to the chosen threshold. For each year from 2013 to 2018, the area covering the Namizimu forest reserve was classified in disturbed and undisturbed pixels using Landsat-8 images at 30m resolution collected at the end of the wet season (Neil, 2018). The yearly disturbance index maps were then combined over the chosen time period to obtain an overall disturbance index according to the number of years in which the pixel was disturbed (Table 1). All the GIS processing was done using ArcGIS (Version 10.6.1).

Table 1 - Disturbance index classification

Number of years of disturbance	Disturbance category
0	Intact
1 to 2	Low disturbance
3 to 4	Medium disturbance
5 to 6	High disturbance

We employed the overall disturbance index to evaluate (a) spatial patterns of degradation in the harvesting area and (b) whether patterns of degradation were associated with human presence by performing proximity analysis (see Appendix 2).

Identify the forest catchment area for the reference population chosen

We defined the boundary of the total forest area used for the supply mapping based on a walking distance approach (Albers and Robinson, 2013; Uchida and Nelson, 2008). The basic assumption of this approach is that the walking distance between villages and harvesting locations in the forest

represents a cost for the individual who collects forest products (Schaafsma et al., 2012). The higher the distance between the village, the higher the cost for the users to reach it and therefore less likely that the individual will harvest in that specific location.

Therefore, to define the forest catchment area we produced an accessibility map that assigns a foot-based travel time to each possible harvesting location (grid cell 30m x 30m) in Namizimu forest reserve considering the 110 selected villages as starting points. The travel time by foot was assumed to be 4 km/h on flat terrain and it is weighted by the slope expressed in gradients, extracted from the 30 arc-second elevation product SRTM30, following the van Wagtendonk and Benedict (1980) formula.[2] We defined the forest area of analysis (called harvesting area henceforth) by including all harvesting locations that can be reached by walking a maximum of 180 minutes. The threshold of 180 minutes was chosen according to the household survey data on walking time to collect forest products (mean=76 minutes, mode=120 minutes; 96% of the sampled households declare to walk 180 minutes or less). To account for the fact that the population of the villages located on the west side of the forest would also harvest forest products from Namizimu forest, we calculated a travel time map for both the west and the east side villages and assigned the pixels that overlay to the group of villages (east or west) for which the walking time is lower. We used cost distance analysis tools to produce the final harvesting area map for the east side villages which covers 31,752 ha of the Namizimu forest reserve.

Mapping the ES potential supply

We quantified the current biomass stock, i.e., potential supply of ES, in the Namizimu forest reserve using a combination of remote sensing data, i.e., Normalized Difference Vegetation Index mapped at 30m resolution, and forest biomass measurements performed on the ground. We thereby produced a map of biomass supply expressed in tonnes, i.e., 1000 kg of dry biomass, at 30 m resolution for the Namizimu forest reserve area.

The field inventory data were extracted from a forest biomass survey conducted in Malawi in 2011 by the Forest Research Institute of Malawi (FRIM). The biomass survey was part of the Forest Preservation Programme funded by the government of Japan which aimed to develop a monitoring, reporting and verification (MRV) system for the reduction of carbon emission in the forestry sector (Makungwa, 2012). The survey covered 17 priority forest reserves across the three regions of Malawi which were selected by the Department of Forestry as pilot sites for the implementation of REDD+ programmes. For the Namizimu forest reserve, 53 sample plots were randomly selected through overlaying grid points at a 4km interval on the whole forest area. Each grid point was associated with GPS coordinates which were used to determine the centre of a circular plot of a size of 0.1 ha. For each plot, all trees with diameter at breast height (DBH) greater than or equal to 20 cm were included in the dataset, the trees species names were recorded and the DBH, tree height (HT), clear length and crown diameter were measured.

The above-ground dry biomass (AGB) of each sampled tree in the plots was estimated using the allometric biomass model (eq. 1) developed for miombo woodlands in Malawi (Kachamba et al., 2016). The carbon content is assumed to be 47% of the dry biomass (Kachamba et al., 2016).

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[2] $V = V_0 e^{-3s}$ where V is the foot-based speed over the sloping terrain, V_0 the base speed of travel over flat terrain (4 km/h in this case) and s is the slope expressed in gradient

310 $$AGB = 0.103685 * DBH^{1.921719} * HT^{0.844561} \tag{1}$$

311 The total AGB value for each plot was calculated by summing up the AGB estimates for each tree in
312 the plot.

313 Next, we estimated a regression model to quantify the relationship between AGB estimates for each
314 plot and the Normalized Difference Vegetation Index (NDVI) georeferenced values to interpolate the
315 biomass value stock across the whole Namizimu area and produce a map of total biomass supply.
316 We used a map of NDVI at 30m resolution as processed by Neil (2018) who used Landsat-8 images
317 collected at the end of the wet season for 2013. The year 2013 was chosen because the images were
318 cloud-free over the Namizimu area, and therefore less prone to measurement error. The NDVI is a
319 spectral vegetation index which is derived from remote sensing images by combining algebraically
320 the values of the spectral reflectance in the red and in the near infrared (Beck et al., 2011). The NDVI
321 index ranges between 0 and 1, where a very low value corresponds to bare areas while high values
322 indicate a high prevalence of vegetation. The spectral index was associated to biophysical variables
323 such as Leaf Area Index, biomass and productivity (Beck et al., 2011; Gizachew et al., 2016).

324 To link the AGB data with the NDVI data, we first had to spatialize the AGB estimates. We did this by
325 creating a circular buffer centred on the UTM coordinates of each field plot, using the plot radius
326 reported in the field measurement dataset. We then overlaid the AGB map with the NDVI map and
327 calculated the average NDVI value for each plot to be used as independent variable in the regression
328 model. We masked the NDVI map using the FAO dominant landscape map at 30 arc-second
329 resolution (Latham et al., 2014) and selected the area classified as forest; this allowed us to exclude
330 areas which are included within the reserve boundaries but that have been deforested (e.g. tea
331 plantations areas). We explored the relationship between AGB and NDVI statistically by calculating
332 the Spearman's rank correlation coefficient (ρ) and visually by analysing the scatter plot of AGB
333 against NDVI. The scatter plot (see Appendix 1) suggested a linear relationship, so we estimated a
334 linear regression model and used the Bonferroni outlier test together with visual analysis of the
335 scatter plot to detect and exclude outliers (Fox, 2016). We then interpolated the biomass estimates
336 across the whole Namizimu forest area using the estimated regression model and assigned value
337 zero to each pixel that had a negative biomass value. For the linear regression model, we excluded
338 four plots from the forest inventory sample because there were no data recorded and we further
339 excluded two other plots because they were identified as outliers using the Bonferroni outlier test.
340 The dataset used for the regression analysis hence included the total AGB estimated for 47 plots of
341 0.1 ha; for each plot there are on average 9 trees (SD 5 trees) and the mean estimated AGB is 6.49
342 tonnes of dry biomass (SD 4.03; mean biomass per ha 62 tonnes).

343 *Current use*
344 To estimate the current demand for dry biomass by the 110 villages of interest, we used the data
345 obtained through a household survey administered in six villages selected from the east side villages.
346 The six villages were selected using a random spatial sampling strategy. The households to be
347 interviewed within each village were selected using a stratified random sampling strategy based on
348 gender, age, and wealth status.

349 We quantified the woody biomass extracted by households based on the sample median yearly
350 consumption of four different forest products: firewood, poles, timber, and charcoal. The quantities
351 extracted for each product were recorded using customary measurement units, such as headloads
352 for firewood. These were translated into dry biomass expressed in tonnes using conversion factors

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adopted by FRIM, available in the literature (Marge, 2009) or information provided in the original survey. The data were collected within a 3-week period. The data collected through household surveys suffer from some limitations which are common in data collection on natural resource uses (Schaafsma et al., 2012). The survey question required respondents to recall how much forest products they collected over a certain time period, and this recalled quantity may be inaccurate. Moreover, collecting forest products from Namizimu forest reserve is illegal unless licensed by the forest department, but some harvesting is done without a license and therefore classified as illegal. Even though we put great care in guaranteeing anonymity to our respondents and obtained information on illegal harvesting from some respondents, it is likely that our data are characterised by underreporting especially for poles and timber collection.

2.3.2. Step 2: model the spatial connection between demand and supply by connecting villages with the forest catchment area

To model the spatial connection between potential supply for ES, biomass available in the forest, and potential demand for ES, forest products consumption of population in the surrounding villages, we simulated a hypothetical co-management scheme. To simulate such scheme, we grouped each village included in the east side villages together with its closest villages by using cluster analysis and assigned to each of these group of villages a sub-area of the forest catchment area, i.e., co-management block, for harvesting activities by using the walking distance approach. We consider each group of villages as the basic unit of analysis for quantifying the potential demand of ES while each co-management block is the basic unit of analysis for potential supply of ES.

We grouped single villages together in a group village, called village cluster henceforth, using a cluster algorithm that aims to minimise the distance between village points within the group and maximise the distance between village points across the different groups. The underlying assumption is that villages which are closer to each other are all close to the same forest sub-area and are also more likely to have aligned interests regarding management objectives. We used the Partitioning Around Medoids (PAM) clustering method which, given the chosen number of clusters (k), finds the optimal partition of the objects contained in the dataset in k clusters (Kaufman et al., 1987). We chose the number of k clusters assuming that each forest management block would be allocated to a group of 5 villages, following interviews with experts from the forest department; this means that the total of 110 villages had to be clustered into 22 groups. The algorithm employed for the clustering exercise (PAM) is usually sub-optimal and the clusters produced by it may vary depending on which are the chosen k starting points from the dataset (Fowler et al., 1981). Therefore, to evaluate the sensitivity of our modelling approach we ran the algorithm several times specifying a different random set of k starting points and verified that the algorithm produces a stable set of clusters, i.e., either the same exact clusters or with very minimal differences (just one village gets allocated to a different cluster).

Next, we defined the harvesting area for each cluster of villages using a walking distance approach. We produced an accessibility map that assigns a foot-based travel time to each possible harvesting location for each village (see Appendix 3 for an example of map output). Then, we produced a single travel time map for each village cluster by taking the average values of the travel maps relative to each village within the cluster. We defined the cluster harvesting area by including all harvesting locations that can be reached within 180 minutes. Finally, we assigned each harvesting location exclusively to one cluster by assigning it either to the cluster that can reach the location or, when the

396 harvesting location falls into two or more cluster harvesting areas, to the cluster that can reach it in
397 the lowest walking time.

2.3.3. Step 3: scenario analysis

Business as usual scenario

400 To evaluate the future ecological sustainability of current management practices and consumption
401 patterns we estimated a business-as-usual scenario to compare quantity of biomass supplied and
402 biomass demanded for each co-management block. The base year is 2015 and we analysed the
403 scenario over a horizon of 15 years, until 2030, with yearly steps assuming an annual population
404 increase and that per household consumption of forest products across time is constant.

405 We calculated the biomass supplied by each block for each year by adding to the current estimated
406 biomass the mean annual increment per ha estimated by Green et al. (2016) for Malawi miombo
407 woodlands. We calculated the total biomass demanded by each village cluster by estimating average
408 consumption of a household based on data collected through household's survey data for six
409 villages. The number of people collecting the various forest products did not differ significantly
410 across the villages (P-value > 0.10 for each forest product, Fisher's exact test), therefore we
411 considered our estimated demand representative of the whole sub-population located in the east
412 side of the forest reserve. Under this assumption we multiplied the estimated amount for the total
413 population of each village cluster of the east side villages to obtain total consumption for each
414 village clusters.

415 We estimated the total population of each cluster using the gridded population map of Malawi for
416 2015 version 2.0 produced by WorldPop (Tatem, 2017) at a 100m resolution, which reports the total
417 number of people for each pixel. We used the modelled gridded population because it is more
418 representative of the spatial distribution of the population across the area than the administrative
419 unit counts. We used the Thiessen polygon tool in ArcGIS, from the proximity toolset, to partition
420 the population map in unique polygons with one village point at its centre. The population of each
421 cluster was calculated as the sum of the population within all villages' polygons of the cluster.
422 Finally, we calculated the number of households assuming a mean household size of four members
423 according to our household survey data. Finally, we estimated the population for each year using the
424 UN world population average exponential rate of growth for Malawi estimated over a period of 5
425 years of 2.66% (medium fertility variant, UN DESA, 2019).

Co-management scenario

427 To evaluate co-management policies and their distributional rules, we performed a monetary
428 valuation exercise using a DCE. We developed a co-management scenario based on the assumption
429 that the quantity of biomass that can be extracted from each co-management block would be
430 capped to its sustainable level. The ecologically sustainable amount is defined as the quantity of
431 biomass that can be extracted without reducing the productive capacity of the forest (Luckert and
432 Williamson, 2005). We assumed that the annual sustainable harvest rate for each forest co-
433 management block is equal to 80% of the AGB growth (Otuoma et al., 2011) and we approximated
434 the AGB growth using the mean annual increment per ha modelled by Green et al. (2016). The
435 average estimated quantity of biomass that Miombo woodlands in Malawi can provide in a year is
436 1.639 t/ha. We calculated the total quantity of biomass that can be extracted under a sustainable
437 management regime by multiplying each block area by the estimated mean annual increment.

438   Finally, we developed a set of hypothetical distributional rules to allocate the sustainable biomass
439   amount available to each cluster to its total population. We converted the total biomass supply
440   expressed in tonnes in number of trees using an average quantity of biomass per tree calculated
441   from our field inventory dataset. We analysed nine different distributional scenarios where the
442   available biomass is distributed in different proportions among better-off ("rich") and worse-off
443   ("poor") village members in each cluster (Table 2). We calculated the total number of rich and poor
444   households using the proportion of people that define themselves as relatively rich and relatively
445   poor in our household sample (31% and 69% respectively) and we assume that this proportion
446   remains constant over time.

Table 2 - Distributional rule for scenarios

	Rich	Poor
	% of total trees	% of total trees
Scenario 1	50%	50%
Scenario 2	10%	90%
Scenario 3	20%	80%
Scenario 4	30%	70%
Scenario 5	40%	60%
Scenario 6	60%	40%
Scenario 7	70%	30%
Scenario 8	80%	20%
Scenario 9	90%	10%

447   The maximum number of trees available for each household under all scenarios is capped to 15 per
448   year as this was the maximum amount that could be harvested in the hypothetical policy scenario in
449   the DCE. To account for variability in forest users' needs we applied our distributional scenarios
450   considering that total biomass is distributed to: (1) the total population, (2) the proportion of people
451   who stated to be interested in harvesting forest products in our sample (97%), (3) the proportion of
452   people interested in harvesting poles and timber (55%) and (4) the proportion of people interested
453   in harvesting timber (21%) (see Appendix 2 for a summary of the results for scenarios (2), (3) and
454   (4)).

455   Finally, we assessed the welfare impact of co-management policies by using WTP estimates from our
456   DCE survey. Our DCE survey assessed the contribution to individual welfare of different levels of
457   biomass provision under a co-management policy and of the distributional rules used to allocate the
458   biomass to other village members identified based on a wealth status category, i.e., rich, and poor
459   village members.

460   In our DCE respondents were asked to choose among co-management options described as a
461   hypothetical distribution of biomass, expressed as the number of trees for each household, among
462   the respondent himself, a village member poorer than them and a village member richer than them.
463   Formally, the individual is assumed to maximise their utility function, i.e., an additive function
464   including all the different characteristics of the co-management policies on offer:

$$U = ASC_j + \beta_{ts}Trees_{self} + \beta_w Water + \beta_{tp}Trees_{poor} + \beta_{tr}Trees_{rich} + \beta_p Price \qquad (2)$$

The marginal WTP for each of the DCE attributes (MWTP) provides a monetary measure of individual's welfare gain (or loss) for a one unit increase of biomass, here described using trees as a proxy. The MWTP associated with allocating one additional tree to a village member poorer than the respondent is calculated as the ratio of the biomass (trees for self) attribute coefficient and the price attribute coefficient:

$$MWTP_{tp} = -\beta_{tp}/\beta_p \qquad (3)$$

We used the estimated MWTP for the biomass attributes to calculate the monetary welfare changes associated with the different distributional scenarios. Water availability is assumed to be constant across the scenarios, primarily because a suitable hydrological model coupled with forest cover was unavailable for the type of biome of the Namizimu forest.

The individual WTP values for a poor and a rich village member for cluster 1 are calculated as:

$$\mathrm{MWTP}_{P\_1} = \mathrm{MWTP}_{self} * \mathrm{Trees}_{poor\_1} + \mathrm{MWTP}_{tp} * \mathrm{Trees}_{poor\_1} + \mathrm{MWTP}_{tr} * \mathrm{Trees}_{rich\_1} \quad (4)$$

$$\mathrm{MWTP}_{R\_1} = \mathrm{MWTP}_{self} * \mathrm{Trees}_{rich\_1} + \mathrm{MWTP}_{tp} * \mathrm{Trees}_{poor\_1} + \mathrm{MWTP}_{tr} * \mathrm{Trees}_{rich\_1} \quad (5)$$

where Trees_{rich_1} and Trees_{poor_1} are the number of trees allocated to rich and poor village members in cluster 1 following the rules of the distributional scenarios. MWTP_{self} does not differ between rich and poor respondents (Dreoni, 2019).

According to the utilitarian framework, the societal welfare value, i.e., the welfare effects of a policy change for the whole population, is obtained as the sum of the individual WTP. The total WTP aggregated over the population for cluster 1 is calculated as:

$$WTP_{TOT\_1} = \mathrm{MWTP}_{P\_1} * Pop_{poor\_1} + \mathrm{MWTP}_{R\_1} * Pop_{rich\_1} \qquad (6)$$

We used total WTP estimates at cluster level to evaluate which distributional scenario maximises total societal welfare, the societal welfare of the poor and of the rich population, for each cluster and for the total forest catchment area, where the latter is calculated as the sum of the total WTP of all clusters.

3. Results

3.1. Potential biomass supply and demand

3.1.1. Current potential supply

The linear relationship between estimated AGB and NDVI is positive and statistically significant (Table 3). We used this model to produce a biomass supply map for the year 2013 (Figure 3). Following Gizachew et al. (2016) we evaluated the model performance using the root mean square error (RMSE) expressed as the percentage of the mean AGB estimated from the field inventory and the absolute bias. The RMSE% is 53% and the absolute bias is 0.0001 t/ha, suggesting that the model's accuracy is comparable with previous work that used a similar approach (Gizachew et al., 2016).

Table 1 - OLS regression results. The dependent variable is the total biomass expressed in tonnes for each 0.1 ha plots regressed on average NDVI for the same plot area.

Intercept	-7.1	**
	(3.523)	
NDVI	19.5	***
	(5.0)	
Observations	47	
Adjusted R2	0.26	
Spearman's rank correlation coefficient (ρ)	0.53	***

Notes: *p<0.1; **p<0.05; ***p<0.01. Standard errors in parentheses.

3.1.2. Current use

The estimated median current consumption of forest products in terms of quantities harvested per household per year is presented in Table 4, along with the relative frequency of households harvesting trees for those products disaggregated at village level. Overall, about three out of every four households collect firewood from Namizimu forest reserve while 13% of our respondents collect poles (Table 4). Tree harvesting for timber and charcoal production is done by 1% and 5% of our sampled households, respectively. The estimated median quantity of dry biomass extracted for firewood is 4.8 tonnes a year, which is very similar to the estimated annual consumption of firewood (4.6 tonnes) for communities living in the surrounding of Chimaliro and Liwonde forest reserve (Jumbe and Angelsen, 2011).

Table 2 - Descriptive statistics of current biomass demand disaggregated by village and product

	Median yearly quantity (t)	Relative (absolute) frequency of households harvesting trees						
		All villages	Village 1	Village 2	Village 3	Village 4	Village 5	Village 6
Firewood	4.8	76%	70%	85%	63%	80%	83%	70%
		(218)	(38)	(45)	(22)	(43)	(35)	(35)
Poles	4.4	13%	11%	23%	6%	9%	10%	16%
		(37)	(6)	(12)	(2)	(5)	(4)	(8)
Timber	6.3	1%	0%	0%	6%	0%	2%	2%
		(4)	(0)	(0)	(2)	(0)	(1)	(1)
Charcoal	8	5%	9%	4%	3%	4%	5%	4%
		(14)	(5)	(2)	(1)	(2)	(2)	(2)
Total	23.5	(288)	(54)	(53)	(35)	(54)	(42)	(50)
Distance from forest (m.)			1,083	223	293	3,085	379	3,669
Distance from major centres (m.)			4,496	14,894	11,373	6,777	8,079	17,825

Notes: Number in parentheses is the absolute number of respondents in each category.

511 The forest products collected by the sampled households are used almost exclusively for domestic
512 consumption, firewood and charcoal are used as main energy source while poles and timber are
513 used as construction material. Very few households engage in trade or other income-generating
514 activities using forest products extracted from Namizimu. We further investigated the use of forest
515 products for income-generating activities among the households interviewed and we found that
516 trade and sales of forest products is very limited in the area, where about 20% of the respondents
517 declare to sell firewood for income when needed, while less than 1% sells poles and timber.

3.2. Spatial connection between demand and supply: village clusters and co-management blocks

520 The distance-based processing generated 22 village clusters with a cluster composed of five villages
521 on average (SD 2). Six clusters (28 villages) were not allocated a forest harvesting area because their
522 entire harvesting area always overlaid another cluster's area with a lower cost distance, and
523 therefore were excluded from further analysis. The average distance from the forest edge of the
524 excluded villages was 3,568 m, compared to 1,437 m for the remaining villages and given the
525 importance of accessibility for harvesting (Albers and Robinson, 2013), excluding the six furthest
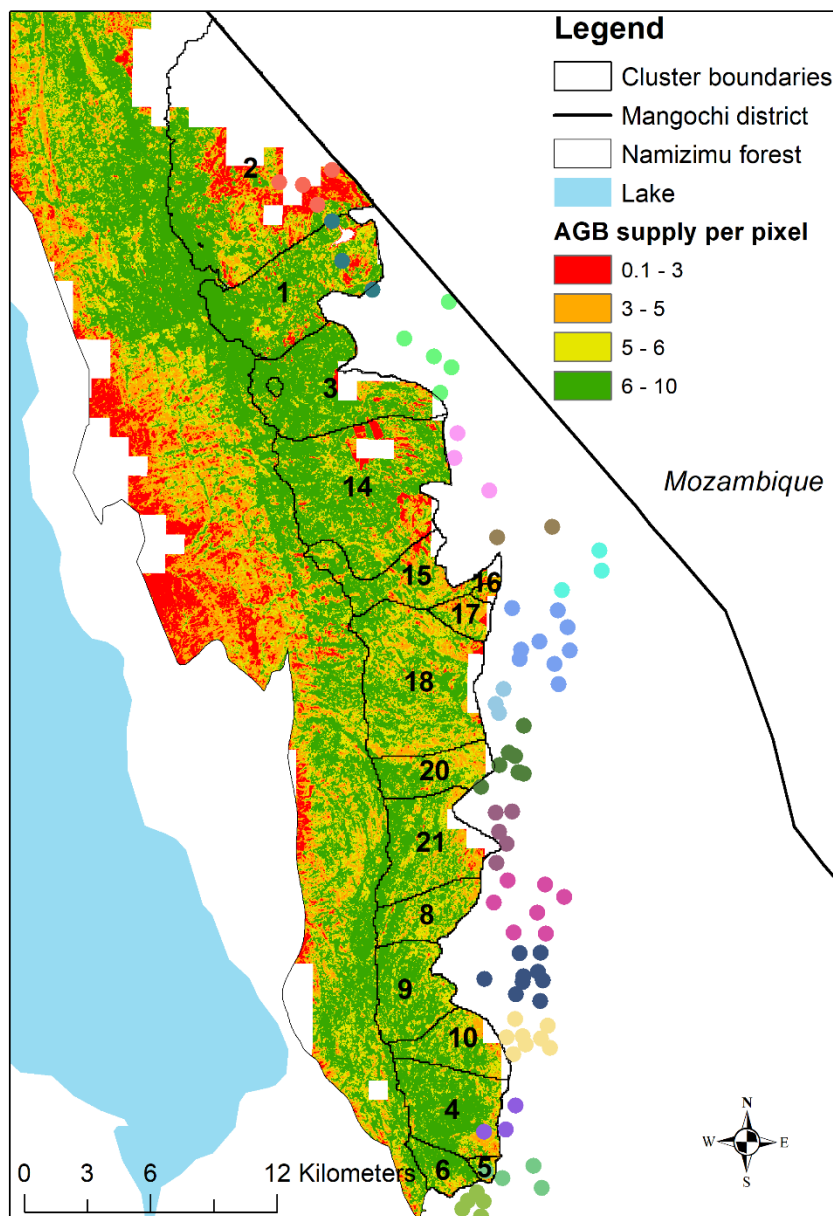526 clusters seems reasonable.

Figure 3 – In the figure we show the results of the potential supply quantification, i.e., the potential biomass that can be extracted from Namizimu forest reserve. The map presents the AGB estimated for each pixel of 30m x 30m size (0.09 ha) and therefore the order of magnitude of the estimates differs from the standard measure of biomass expressed in t/ha. The biomass map is overlaid with the boundaries of the modelled co-management blocks. These blocks together form the forest catchment area corresponding to the cluster of villages located on the east side of the forest. Villages of the same colour belong to the same cluster and are allocated one specific co-management block in the forest.

The average size of the co-management blocks is 1,984 ha but varies greatly across clusters (SD 1,821 ha – Table 5 and Figure 3). As discussed above, the delineation of co-management areas is

18

negotiated on a case-by-case basis according to multiple criteria, therefore there are no standards against which we can compare our simulated co-management blocks. However, actual co-management policies implemented in Liwonde forest reserve located in southern Malawi (area of 26,991 ha) show similar variability in block forest size ranging from 357 ha to 2721 ha. The total above-ground biomass supplied by each co-management block expressed in t/ha ranges between 31.15 and 69.47. The range translated in carbon content (14.64-32.65 tC/ha) is in line with the range of 13-30 tC/ha estimated by Shirima et al. (2011) for a Miombo area in Tanzania and the average of 21.2 tC/ha estimated by Ryan et al. (2011) for a Miombo area in Mozambique.

Table 3 - Current situation in the base year (2015) for each cluster

Cluster n.	Number of villages	Co-management block size (ha)	Population (HH)	Biomass (t/ha)	Available number of trees/HH	Current consumption sustainable?
1	3	3,070	148	66.70	56	YES
2	4	6,957	895	31.15	21	YES
3	5	2,738	1,028	63.80	7	NO
4	3	1,804	193	68.33	25	YES
5	4	120	434	59.07	1	NO
6	7	511	1,165	68.38	1	NO
8	7	1,003	1,563	65.65	2	NO
9	9	1,430	1,114	69.47	3	NO
10	8	1,093	1,840	58.57	2	NO
14	3	4,381	599	61.29	20	YES
15	2	1,506	789	57.35	5	NO
16	3	93	1,135	45.32	0	NO
17	9	420	2,394	52.80	0	NO
18	3	3,613	362	61.83	27	YES
20	7	1,042	706	62.23	4	NO
21	5	1,969	738	59.79	7	NO
Total	**82**	**31,752**	**15,102**	**59.48**		
Excluded clusters	**28**		**5,964**			

Notes: clusters n. 7, 11, 12, 13, 19 and 22 are the clusters excluded as they were not allocated any area.

The number of households (HH) assigned to each co-management block is also characterised by great variability ranging from 148 households in cluster n. 1 to 2,394 households in cluster n. 17. Such spatial variability determines a different initial distribution of forest resources to each household depending on the size and the ecological status of each co-management block. The last column reports the absolute number of trees that can be sustainably harvested by each household according to their co-management block size and shows that there are great inequalities across co-management blocks in the amount of biomass that can be extracted. Specifically, cluster n. 1, 2, 4, 14 and 18 can potentially extract a high quantity of biomass and would certainly be able to satisfy average firewood demand for each household, estimated at 10 trees. The households assigned to the clusters n. 5, 6, 16 and 17 are instead in great disadvantage given the small size of their co-management block. The remaining clusters are also characterised by a low number of trees that

557　could be harvested sustainably by each household implying that not everyone would be able to
558　obtain the current demanded quantities.

3.3. Scenario analysis

3.3.1. Business as usual scenario

561　The current initial stock of biomass is sufficient to cover demand for all clusters however the current
562　yearly consumption is higher than the capacity of the forest to re-generate in many clusters. The
563　analysis of changes over time (see Figure 4) shows that the current consumption patterns are
564　unsustainable and that the forest harvesting area would be highly degraded by 2030. The average
565　amount of biomass provided per ha in 2015 is 59.48 t/ha, decreasing to about 31.40 t/ha by 2030,
566　assuming that all households continue harvesting and, when their co-management block is
567　completely degraded, either harvest from a different co-management block or start trading with
568　other areas. Figure 4 shows that the number of blocks that are fully degraded and deforested and do
569　not provide above ground tree biomass, represented as white blocks, increases over time. It further
570　shows that the co-management blocks located in the south area are more prone to degradation due
571　to the higher population density and a lower availability of forest ha per person. In the areas located
572　in the north, the biomass supply is stable over time; due to a lower population density the biomass
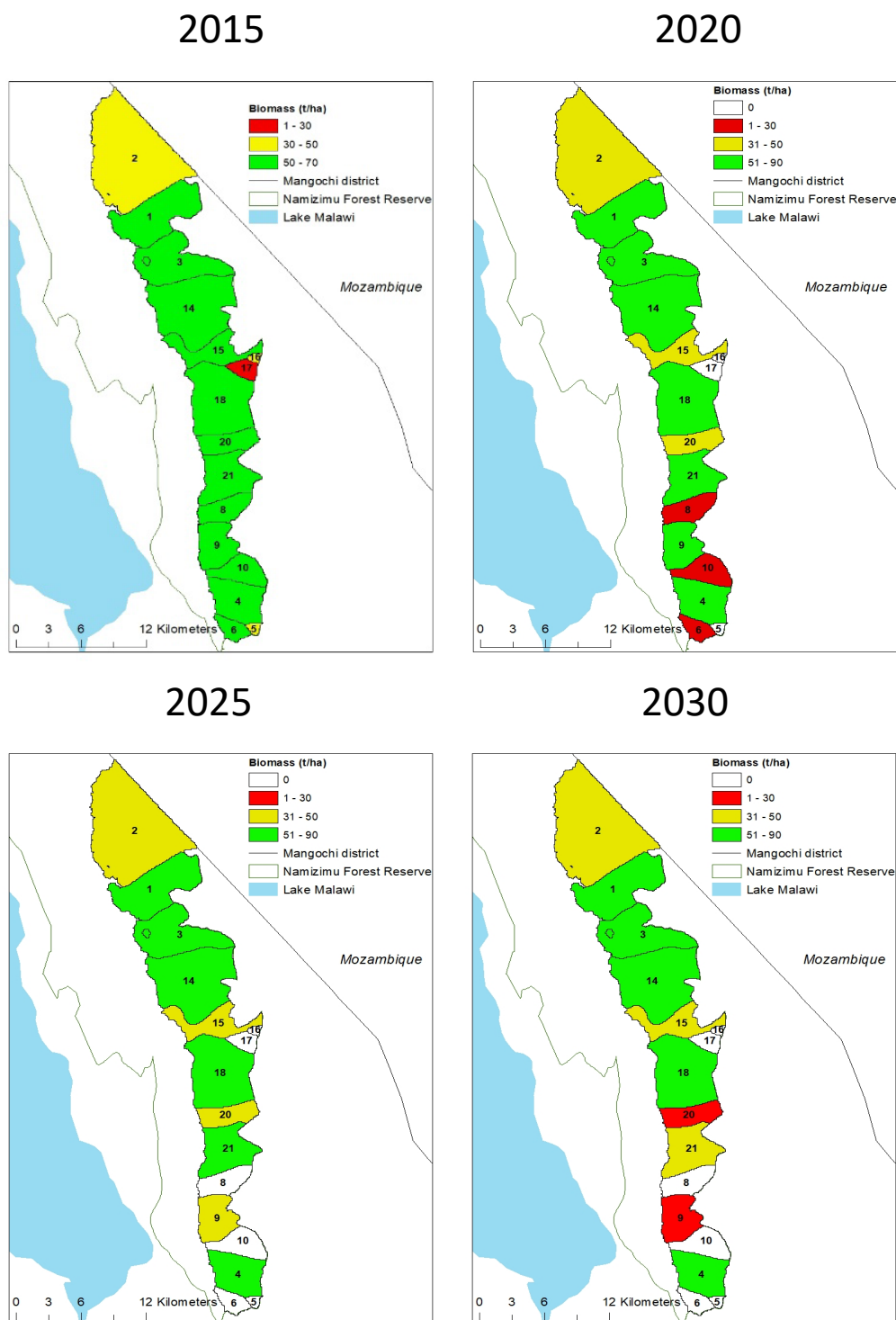573　supply and demand budget is positive, and forest can potentially re-generate.

2015

2020

2025

2030

Figure 4 – Scenario analysis (BAU)

3.3.2. Co-management scenario

In Table 6 we present the average distribution of trees between rich and poor households for each scenario presented in Table 2 and the total WTP for the whole forest area in US$.[3] The scenario that maximises aggregated WTP for the total population is B5, where the rich village households would be allocated about 5 trees, while the poor would get about 4 trees. We also show the total WTP for

579 each social group to provide a monetary measure of welfare trade-offs between the two groups
580 across the different distributional scenarios. The distributional scenario that maximises total welfare
581 (total WTP) is not the same as those scenarios that maximise total WTP for the poor (B2) or the rich
582 group (B9). By measuring welfare with a monetary estimate, WTP, we are able to compare
583 differences among these scenarios and quantify trade-offs. For example, we can verify that
584 implementing scenario B5 produces a loss of welfare of the poor group of about 7% of the maximum
585 welfare that can be obtained with scenario B2, but it produces a gain of 93% in welfare for the rich
586 group compared to the B2 scenario. Hence, moving from B2 scenario to B5 scenario will greatly
587 benefit the welfare of the rich group, while the loss for the poor group would be small in
588 comparison. The gain for the rich group could potentially compensate the loss for the poor group,
589 and therefore from a social welfare perspective overall society is better off with B5 than with B2.
590 However, this type of monetary compensation and transfer would not take place thereby increasing
591 inequality in allocation. Similarly, implementing the distributional scenario that maximise welfare of
592 the rich (scenario B9) produces a loss in welfare of the poor of about 60% compared to the B2
593 scenarios and a loss of total welfare of about 20%. Implementing a scenario that is strongly in favour
594 of the rich would lead to a reduction of welfare for the poor group which is not compensated by the
595 gain in welfare of the rich group.

Table 6 – Average distribution of trees and total WTP estimates across distributional scenarios

Scenarios	Trees/HH (poor)	Trees/HH (rich)	Total WTP	Total WTP (poor)	Total WTP (rich)
	Rich gets more				
B5	4.02	4.91	399,205	264,046	135,159
B6	3.08	6.27	386,089	225,502	160,586
B7	2.36	6.83	359,845	191,035	168,810
B8	1.62	7.24	328,468	154,644	173,824
B9	0.82	7.65	293,003	114,565	178,438
	Poor gets more				
B2	4.94	1.79	355,303	285,558	69,745
B3	4.63	3.36	386,850	283,251	103,599
B4	4.33	4.23	395,991	274,501	121,490
	Equal share				
B1	4.30	4.30	396,312	273,455	122,857

Notes: the scenario that maximises welfare, i.e., higher total monetary value, is in red

596 Moreover, we performed a disaggregated analysis at cluster level to understand the trade-offs
597 across different distributional scenarios when examining them at a more local scale of analysis and
598 therefore considering the differences in ecological endowments for each cluster. Figure 5 shows
599 which distributional scenarios maximise total WTP (blue), WTP of the poor group (red) and WTP of
600 the rich group (green) for each cluster. For the clusters 1, 2, 4, 14, and 18, many scenarios are
601 optimal according to these three criteria, including two scenarios that are considered optimal at

[3] The adjusted exchange rate is 248.38 MKW = 1 US$ PPP (purchasing power parity) using the PPP
conversion factor for private consumption for the year 2018 (World Bank, International Comparison
Program database)

602 aggregate level (B2 and B5). These clusters have a high availability of biomass with an average
603 number of available trees per person of at least 20 trees, where under many distributional rules
604 each village member would be allowed to harvest the maximum number of trees per year (15 trees).
605 For the remaining clusters, where biomass supply is more limited, the B9 scenario is found to
606 maximise total WTP and WTP for the rich group in many cases, while the B2 scenario is found to be
607 optimal for the poor group as at the aggregate level. The figure shows that there is high variability
608 across clusters about what distributional scenarios maximise the three WTP sums. Indeed, the
609 "optimal" distributional scenario depends on the combination of many factors including, among
610 others, the ecological endowment of the forest area examined and the distributional preferences of
611 forest users.
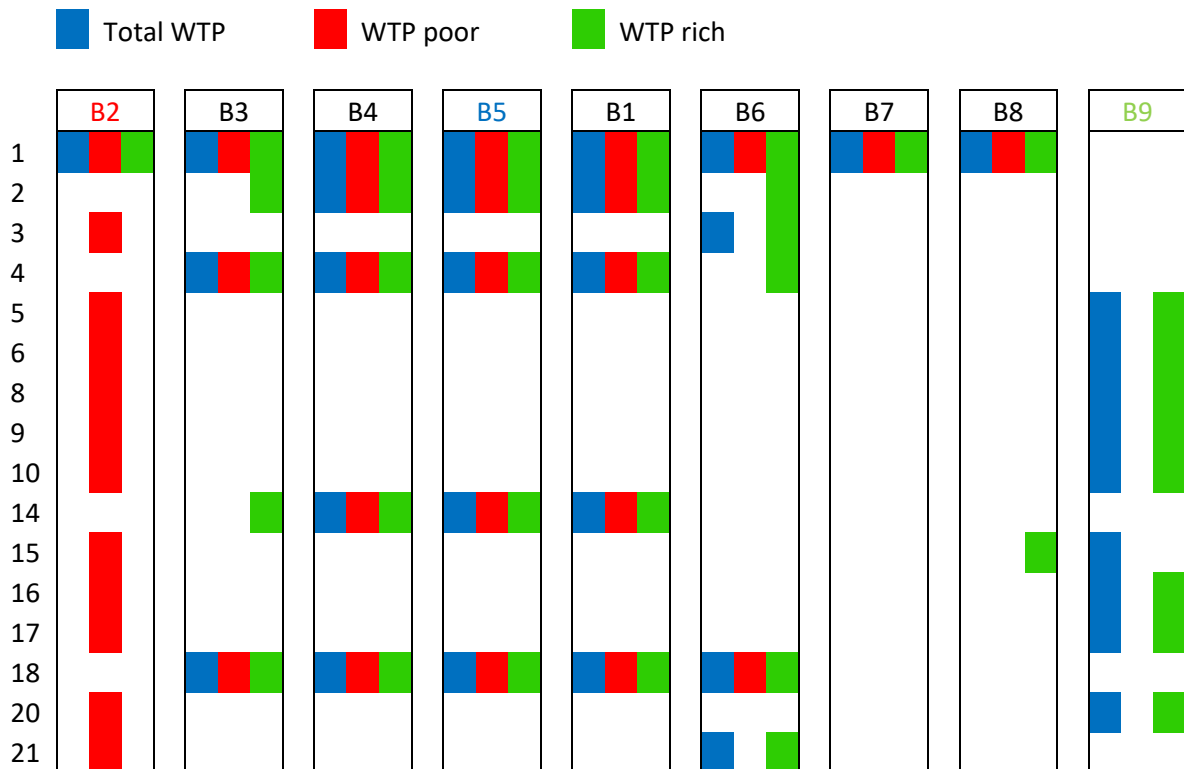
613 Figure 5 – In the figure we show the distributional scenarios that maximise the WTP for the total
614 population, for the poor group and for the rich group for each single cluster. The cluster number is
615 reported on the left, and the cells are coloured if the scenario maximises maximise total WTP (blue),
616 WTP of the poor group (red) and WTP of the rich group (green). Note that scenario B5 maximises
617 total WTP at aggregate level, scenario B2 maximises WTP for the poor group and scenario B9
618 maximises WTP for the rich group.

619 In figure 6 we examine in more detail three clusters which differ in their biomass availability, to
620 assess the welfare trade-offs involved in these three cases: Cluster 2 has a high initial supply of
621 biomass (21 trees per person) while cluster 3 has a medium supply level (7 trees) and cluster 6 a low
622 level (1 tree). Here, the total WTP for both social groups and relative to poor and rich expressed in $
623 is scaled between 0 and 1 using a minmax normalization where 1 corresponds to the scenario that
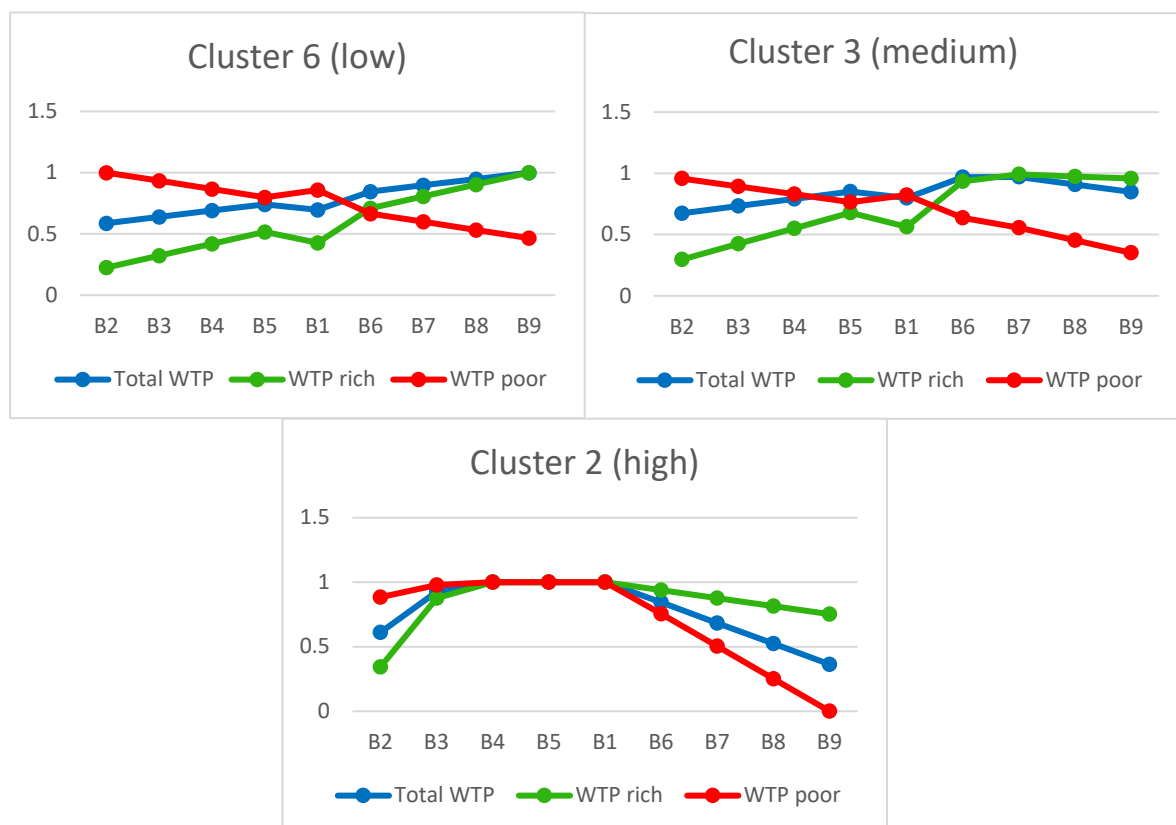624 maximise WTP while 0 corresponds to the scenario with the lowest WTP possible.

Figure 6 - The figure shows minmax normalised WTP values for three selected clusters, n. 2, 3 and 6, to demonstrate how total welfare gains differ across scenarios for clusters with different ecological endowments (low, medium, and high endowment).

625   Figure 6 shows the mean individual WTP for a poor and a rich village member, i.e., their individual
626   welfare gain, and their total WTP. When there is a high availability of biomass (cluster 2), the
627   scenarios B4 and B5, where the poor group gets more biomass, and B1 (equal share) are all optimal
628   for the total population and for both social groups and therefore trade-offs can be minimised. Yet,
629   choosing a distributional rule which further polarizes the distribution of biomass in favour of one of
630   the wealth status groups will also reduce total welfare. For cluster 3 (medium availability of biomass)
631   the optimal scenario in terms of total WTP is the B6 scenario where the rich group gain a higher WTP
632   compared to the poor group but the difference between the two groups is moderate; the B5
633   scenario minimises this difference but lowers total WTP for the whole population compared to B6.
634   Finally, when total biomass available is low, as in cluster 6, the trade-offs between WTP of the poor
635   and the rich group is the most extreme – as can be seen by the opposite trend of the green and the
636   red line in Figure 6 – and the WTP of one wealth status group can only be maximised at the expense
637   of the other.

638   Trade-offs are also reduced if biomass is distributed just to the proportion of people that stated to
639   have interest in harvesting specific forest products as elicited in our household survey (Appendix 2).
640   We carried out this analysis to simulate alternative management arrangement that may arise. We
641   found that the lower the number of people among which forest products are distributed, the higher
642   the convergence of the three criteria toward one scenario, i.e., trade-offs are minimised.

4. Discussion

This paper proposes a methodological approach for the analysis of normative trade-offs between local welfare and forest conservations objectives in forest ecosystem governance. We contribute to the literature by combining a biophysical assessment of ES potential availability and actual flow with a welfare assessment that integrates the impact of distributional outcomes (Chaudhary et al., 2018; Daw et al., 2011; Dawson et al., 2017; Fisher et al., 2014). We quantified the potential supply of provisioning forest ES, i.e., woody biomass, and link it to the current demand of forest-adjacent communities to identify the amount of ES effectively captured by the population. We then assessed the effect on total welfare of the beneficiaries of different hypothetical distributional scenarios arising from a hypothetical co-management policy. We show that, due to the spatial mismatches between demand and supply, the distribution of provisioning ES to the rural population across the whole harvesting area is unequal in biophysical terms (see Table 5). By coupling biophysical modelling with WTP analysis, we then provide an assessment of total welfare gained by the population across different distributional scenarios of biomass. Our WTP analysis captures the effect of distributional rules on welfare at the aggregate level, i.e., for the total population and the whole forest harvesting area, and at disaggregate level, i.e., for the two wealth status groups and for each cluster.

The biophysical assessment reveals that the current patterns of consumption are ecologically unsustainable and demand for ES exceeds sustainable supply in many co-management blocks leading to forest degradation. The disturbance index shows that the areas closer to human settlements have been highly disturbed during the 2013-2018 period. The comparison of supply and demand shows that the distribution of biomass is unequal across the whole harvesting area (see Table 5) and therefore not all demand can be met whilst guaranteeing that forests would re-generate. We found that only in five co-management blocks, which account for 15% of the total population of forest users, the current stock of biomass would be able to support the estimated current biomass demand. Under the business-as-usual scenarios, the current consumption of forest products is unsustainable in most areas, especially in the southern area of the forest, where the population density is higher.

The WTP analysis shows that the distributional scenario that maximises total societal welfare in monetary terms across the whole harvesting area is the scenario that distributes 40% of biomass to the rich group while the remaining 60% is allocated to the poor group (see Table 6). However, this scenario (B5) does not maximise total WTP for all clusters (see Figure 5). Whilst at the aggregate level an (almost) egalitarian distribution of biomass between poor and rich groups is found to be increasing the welfare for the whole population the most, at disaggregated level by cluster a distribution skewed towards the rich maximises total WTP in many clusters where biomass availability is low. However, forest benefits are especially relevant for the poorest village members, and therefore a policy maker could opt to develop institutional mechanisms that maximise the welfare of the poor group on normative grounds, instead of maximising total welfare. In such situations, our analysis shows that the scenario that maximises WTP for the poor group at aggregate level (B2) produces a total WTP almost equivalent to B5, but it also maximises the difference between welfare of the rich and the poor at the aggregate level (see Figure 6).

The WTP analysis implemented in this paper can be used to examine welfare trade-offs between wealth status groups both at the aggregate and disaggregate level using a single indicator (WTP).

Important improvements upon earlier assessments of our methodology is that it considers the contribution to human well-being of the actual ES flow, as opposed to potential, by estimating potential demand and supply and modelling their spatial connection and incorporates spatial heterogeneity in the analysis (Bagstad et al., 2014; La Notte et al., 2017). Moreover, our methodology enables disaggregated assessments which can account for the differential ability of social groups to capture the flow of ES (Daw et al., 2011; Fisher et al., 2014) and evaluate the trade-offs involved in ES governance ex-ante instead of evaluating them ex-post (Chaudhary et al., 2018b; Dawson et al., 2017). As such, our study provides a method for supporting policymakers in choosing the distributional rule that minimize trade-offs between efficiency, i.e., maximizing total welfare, and livelihoods objectives.

Our methodology relies on coupling multiple models used to estimate both biophysical and monetary quantities; such estimations required a set of simplifying assumptions. To quantify the biomass available, we relied on remote sensing images, given that we lacked an updated forest inventory dataset with full coverage of the area, but our estimates are characterised by a relative predictive error of about 50% and uncertainty related to measurement errors. Using SAR (synthetic aperture radar) remote sensing technologies, which overcome the problem of clouds, may improve the predictive ability of the regression model used (Mitchell et al., 2017). Furthermore, to link the potential supply and potential demand, we simulated the spatial connectivity through developing an accessibility map which included walking distance weighted by slope. Such accessibility map could be complemented by other factors deemed as relevant for forest products collection if data were available, such as quality of forest area as a proxy of products availability or distance to market and information on possible substitutes which may increase or reduce the likelihood to harvest from the forest reserve (Albers and Robinson, 2013). Finally, we did not calculate confidence intervals of the aggregate welfare measures so we cannot assess whether the welfare measures across scenarios are statistically significantly different.

5. Conclusion

This paper proposes a combination of methodological approaches for the assessment of normative trade-offs between local welfare and forest conservation objectives in forest ecosystem governance. We choose as a case study area Namizimu forest reserve, located in the Mangochi district in Malawi, where community-based management policies may be implemented in the nearer future to pursue sustainable forest management. The analysis shows how the current estimated level of consumption of forest resources from the local population is ecologically unsustainable. We also show how current estimated consumption patterns may be unsustainable in the future, especially in the south part of the forest area where population density is higher.

We then simulated a hypothetical community-based management scenario to evaluate the welfare impacts of this type of policies. In a co-management policy, the ability of the local population to extract forest products from the forest reserve would be capped to a sustainable level to guarantee the ability of the forest to regenerate. In a spatial analysis we assessed the total quantity of forest resources that could be extracted in a sustainable way from the forest reserve when co-management is implemented. The results show marked differences across space due to differences in forest structure and the ability of the population to access the forest area. Such spatial differences in ecological endowments will determine differential impacts of the policy on the welfare of the local population. It is important to take the spatial variation in resource provision into account when

729   determining the distributional rules for allocating forest resources to the forest-adjacent

730   communities.

731   Finally, we performed a monetary valuation of the welfare impacts of different distributional

732   scenarios under co-management using a stated preference technique. We demonstrated how the

733   results of this monetary valuation can be used to inform decision-making for ex-ante valuation of ES

734   governance policies. In this study, we found that the distributional scenario that maximises total

735   societal welfare for the whole forest area examined follows from the distributional rule where the

736   worse-off village members are allocated a slightly higher amount of biomass (60% of the total) while

737   the better-off village members are allocated the remaining 40%, i.e., a pro-poor distributional rule.

738   While the outcomes of such rules for other areas will depend on the local wealth distribution, an

739   important emerging message is that pro-poor distributions are both ecologically sustainable and

740   locally socially acceptable.

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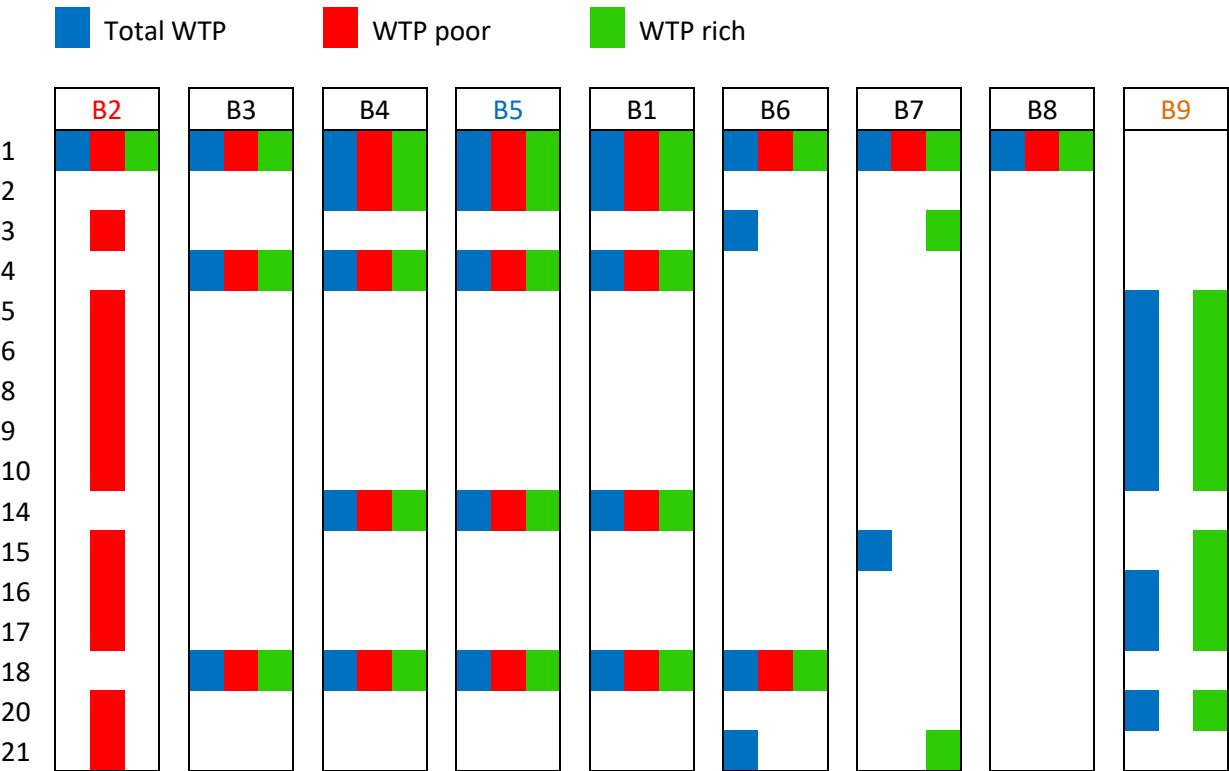963   community forest management and woodfuel policy in Malawi. Energy Policy 38, 3717–3730.

964

Appendix 1 - Scatter plot of AGB against NDVI
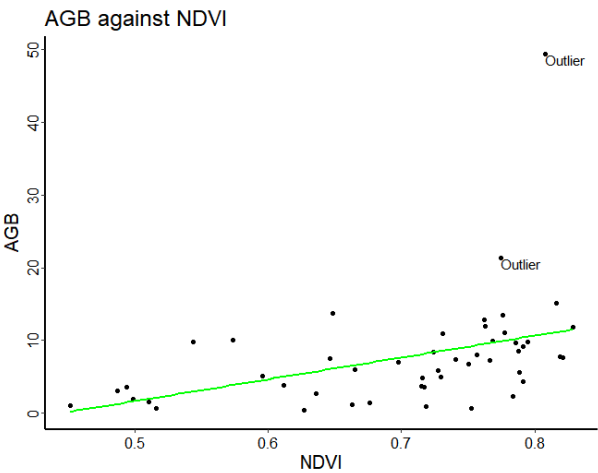
Figure 1 - Scatter plot of AGB against NDVI that indicates that the relationship is linear and that there are two outliers in the dataset.
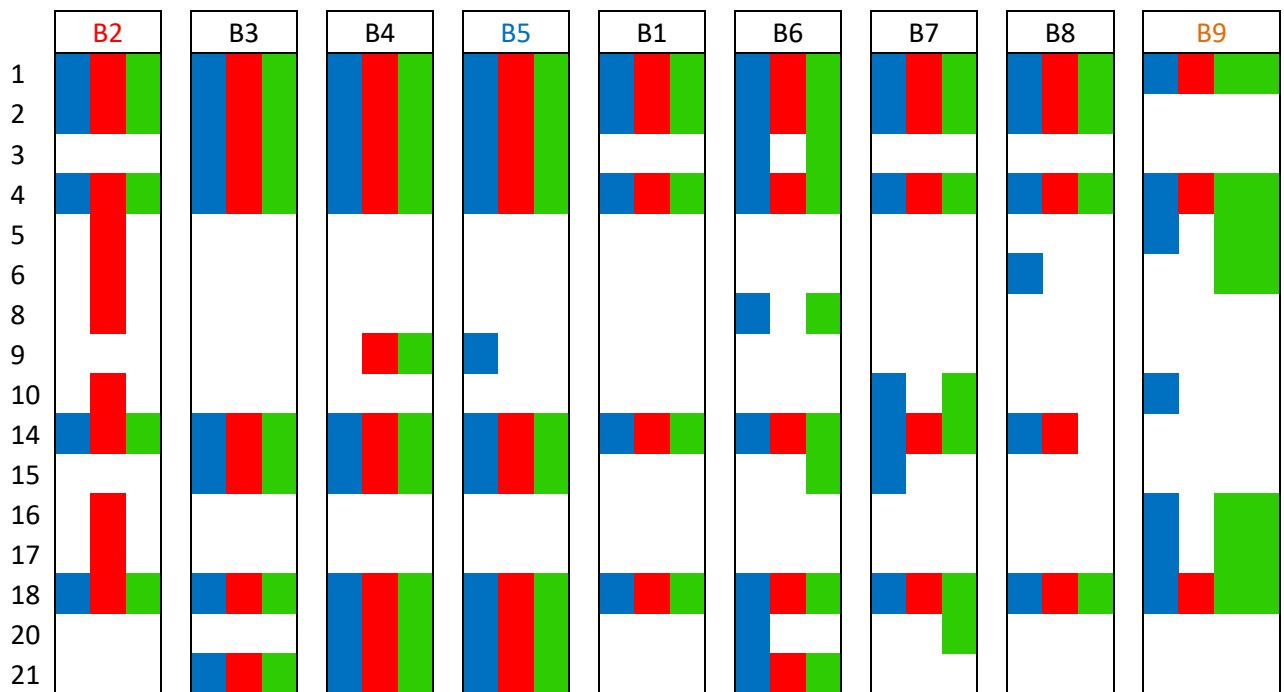
Appendix 2 – Disaggregated analysis at cluster level

Figure 1 - Distributive scenarios that maximise the three WTP criteria across clusters considering that biomass is distributed to the proportion of people who stated to be interested in harvesting forest products (population size 97% of total population)
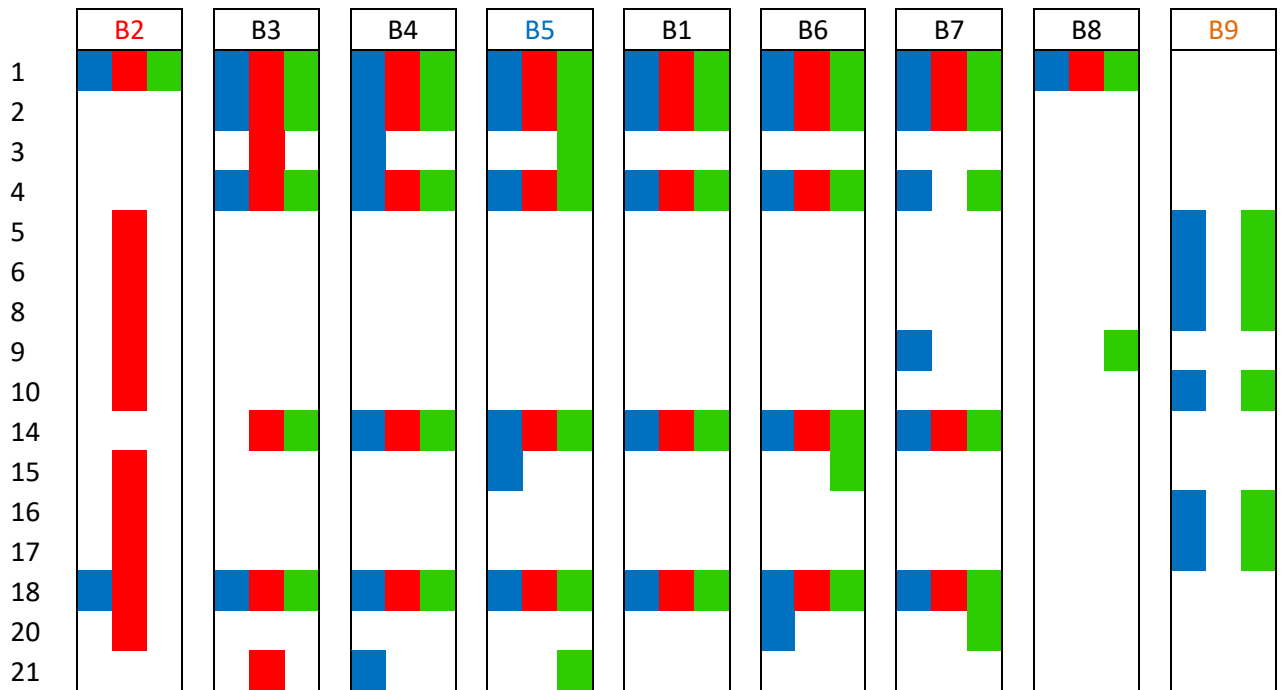
Figure 2 - Distributive scenarios that maximise the three WTP criteria across clusters considering that biomass is distributed to the proportion of people who stated to be interested in the proportion of people interested in harvesting poles and timber (population size 55% of total population)

| Total WTP | WTP poor | WTP rich |

Figure 3 - Distributive scenarios that maximise the three WTP criteria across clusters considering that biomass is distributed to the proportion of people who stated to be interested in harvesting timber (21%)

979 # Appendix 3 – Cost-distance analysis
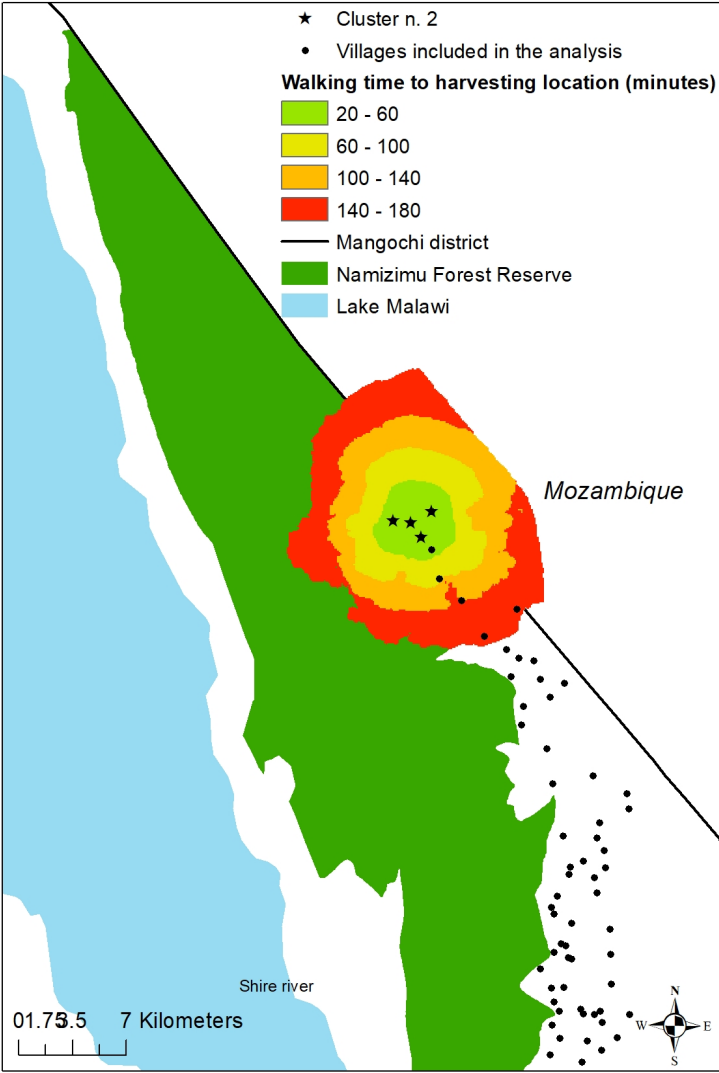
981 Figure 2 – Example of cost distance analysis

982

Appendix 4 - Forest degradation in Namizimu forest reserve

983

984 Figure 2 shows the aggregated disturbance index map for the period 2013-2018 which approximates
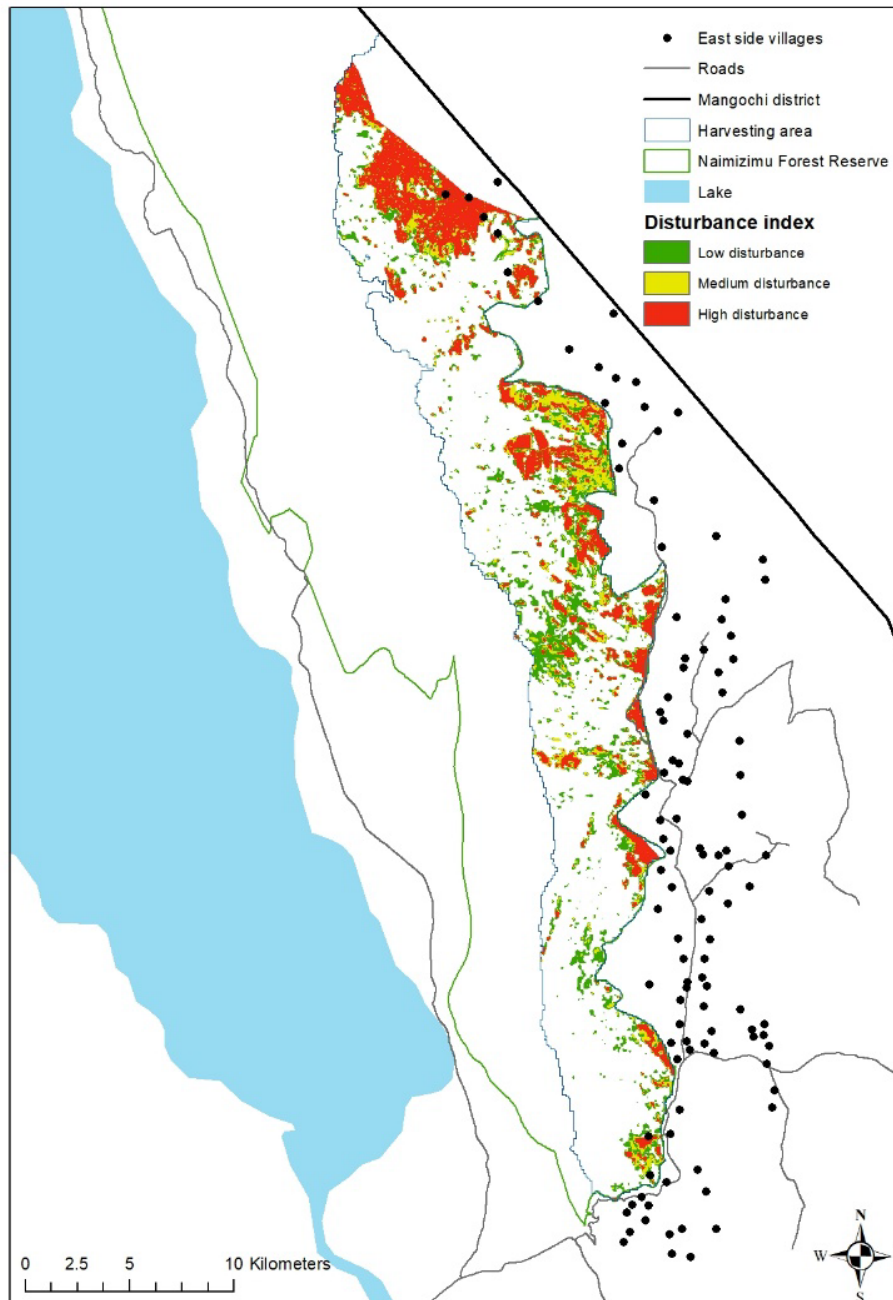985 the intensity of disturbance in the reference area used for the analysis.

987   Figure 1 - Degradation index

988   The pixels classified as highly disturbed are 48% of the total number of disturbed pixels, while the
989   low and medium disturbed ones are respectively 32% and 20%. The average distance from low and
990   medium disturbed pixels to the east side villages is about 3.5 km, while the average value for high
991   disturbed pixels is 2.6 km indicating that areas closest to human population are more likely to be

992   highly degraded. The average biomass per hectare of the highly disturbed areas is 38 tonnes, while
993   the average biomass for the entire harvesting area is 59 tonnes. This shows that the pattern of forest
994   disturbance shown by the index captures a decline in biomass available in the area, confirming that
995   degradation occurs and that the stock of biomass available declined over time especially in areas
996   closest to human settlements.