Effects of forest management and land use on regeneration in REDD+ villages, southeastern Tanzania

by

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Abstract

Tropical deforestation contributes ~17% to global anthropogenic carbon emissions, and is associated with in declines biodiversity, ecosystem services and livelihood benefits. Dryland forests in the miombo region of Africa cover ~3.6 million km^2 and support nearly 100 million people, and are being incorporated into the UN-REDD+ (Reducing Emissions from Deforestation and forest Degradation) efforts to reduce local-scale deforestation and forest degradation. Although reducing deforestation rates is the primary goal of REDD+, fostering rules and regulations that also promote regeneration in degraded areas will play a critical role in mitigating net emissions. This study considers the effects of the institutional structure of forest governance and associated land uses on forest regeneration in REDD+ villages operating under community-based forest management in comparison to centrally-managed forests.

Seedling density was used as a proxy for recruitment and modeled using a generalized linear model with several environmental parameters to test for the effects of forest governance and land use on tree regeneration. The environmental parameters that best predicted recruitment were fire frequency, tree biomass, and soil clay content. Predicted recruitment was significantly higher in community-managed forests (1.11 seedlings m$^{2\ -1}$) than centrally-managed forests (0.63 seedlings m$^{2\ -1}$), supporting previous studies showing community managed areas have healthier forests. Centrally-managed lands had lower predicted recruitment across all lands uses, with the lowest regeneration on centrally-managed timber land (0.50 seedlings m$^{2\ -1}$). This demonstrates that environmental conditions experienced by seedlings are not only effected by land use type, but also the institutional structure that is governing it. Management policies that reduce fire frequencies and retain threshold levels of biomass for shade and seed sources will help facilitate forest regeneration and maintain ecological goals.

1. Introduction

Tropical forests contain 70-80% of the carbon in the terrestrial biosphere (Saatchi *et al* 2011, Baccini *et al* 2012), yet is estimated that around 30% of global forest land has been cleared and 20% has been degraded (WRI 2011). The net emissions from the forestry sector contribute ~17% of anthropogenic carbon emissions (Van der Werf *et al* 2009), making their loss not only a concern for biodiversity conservation, ecosystems services, and resource depletion, but also a critical opportunity for reducing greenhouse gas emissions. Emerging international initiatives such as the United Nations REDD+ (Reducing Emissions from Deforestation and Forest Degradation) program seek to decrease tropical deforestation by incentivizing reduced reliance on forest-based income in rural communities through carbon market financing. The success of both social and ecological goals of these programs depends on mode of governance in which they are applied (Sandbrook *et al* 2010). Community-based forestry management is being used as a model for project implementation (Bond *et al* 2010, MCDI 2010, World Bank 2010) because it gives autonomy and ownership of resources to communities, which is shown to be associated with positive forest outcomes (Bowler *et al* 2012, Ostrom 1990, Porter-Bolland *et al* 2012). For example, carbon storage tends to increase when a community owns its forest commons (Chhatre & Agrawal 2009) and deforestation rates are shown to be lower (Nolte *et al* 2013) because people tend to limit consumption of forest products. This study considers the effects of governance on forest regeneration in REDD+ pilot project sites in southeastern Tanzania.

The importance of appropriate land management and use policies to increase forest productivity and offset carbon dioxide emissions has been recognized in many studies (i.e. Ahrends *et al* 2010, Burgess *et al* 2010, DeFries *et al* 2010, Hurteau *et al* 2013, Knoke *et al* 2012, Rudel *et al* 2004, Post *et al* 2012). While maintaining existing forest biomass has garnered most attention, forest regeneration within disturbed or deforested areas will be necessary to regrow carbon stocks and has been recognized as significant potential source for carbon sequestration (WRI 2011, Post *et al* 2012) and will play a role in REDD+ carbon accounting (Angelsen *et al* 2009; for carbon accounting methods see Goetz *et al* 2009). An estimated 0.6 PgC yr^{-1} could be sequestered globally through improved forest management and forest regeneration (Watson *et al* 2000) in addition to the estimated 1.1-1.6 PgC yr^{-1} already being sequestered (Baccini *et al* 2012, Post *et al* 2011); this amount approaches the gross global emissions released from deforestation and forest degradation of 2.9 ± 0.5 PgC yr^{-1} (Pan *et al* 2011). Yet little research relates how forestry management institutions and land use effect forests regeneration, especially in African forests (Campbell 1996, Martin *et al* 2012).

Though most tropical deforestation avoidance efforts focus on South America and Southeastern Asia, Africa is important because it contains 25% of the tropical forest carbon stock (Saatchi *et al* 2011). The miombo woodlands of Africa are the largest forest type, covering ~3.6 million km^2 and supporting ~100 million people, and pressures for resources result in deforestation rate of ~34,000 km^2 yr^{-1} (FAO 2010). Furthermore, 37% of African humid tropical forest canopy cover has been degraded below 50% (Asner *et al* 2009), and possibly higher degradation in miombo woodland though this is difficult to quantify (Chidumayo & Gumbo 2011). Deforestation and degradation in miombo woodland is driven by small-scale agricultural expansion and the extraction of primary products including selective timber harvest, charcoal production, and non-timber products such as food, fuel wood, and building poles (Fisher 2010, Abbot & Homewood 1999, Ahrends *et al* 2010, Luoga *et al* 2000, Swartz & Caro 2003). Each of these land use activities alter the physical environment experienced by trees, and the institutional arrangements governing the forest vary the human population pressure, resource allocation, and rule enforcement. These four land uses have been suggested as successive levels of resource extraction pressure (Swartz & Caro 2003), and therefore we chose to use land use type nested within the institutional arrangement of forest governance to serve as our treatments.

In this study we explored the forests' ability to regenerate under these institutional arrangements. Since seedlings are the most sensitive life stage to changes in environmental conditions (Silvertown *et al* 1993, Ibáñez *et al* 2007, Nunn *et al* 2005), we chose to use seedling density as a proxy for current recruitment rates and regeneration capacity. Although static population structure data does not necessarily predict population growth rates (Chessen & Huntly 1989, Condit *et al* 1998), seedling population dynamics have been shown to be correlated with adult population trends (De Steven 1994). Generally, decreases in seedling densities lowers the probability of recruitment into the adult stage, and can lead to population bottlenecks and declines in forest regeneration or expansion (Acaio *et al* 2007, Campbell 1996). The minimum numbers of seedlings sufficient to avoid population bottlenecks depends on birth and death rates of seedlings and species-specific growth rates (Condit *et al* 1998). For example, fast growing species with low mortality can have smaller minimum seedling density requirements than slow growing species with mortality rates. Also, environmental factors effect birth, death, and growth rates in combination with species' biological preferences (Chessen & Huntly 1989). Here we combine data across tree species with different life history traits. Though we cannot predict minimum seedling values needed to avoid population bottlenecks, by comparing relative densities across all species we can elicit recruitment response to governance.

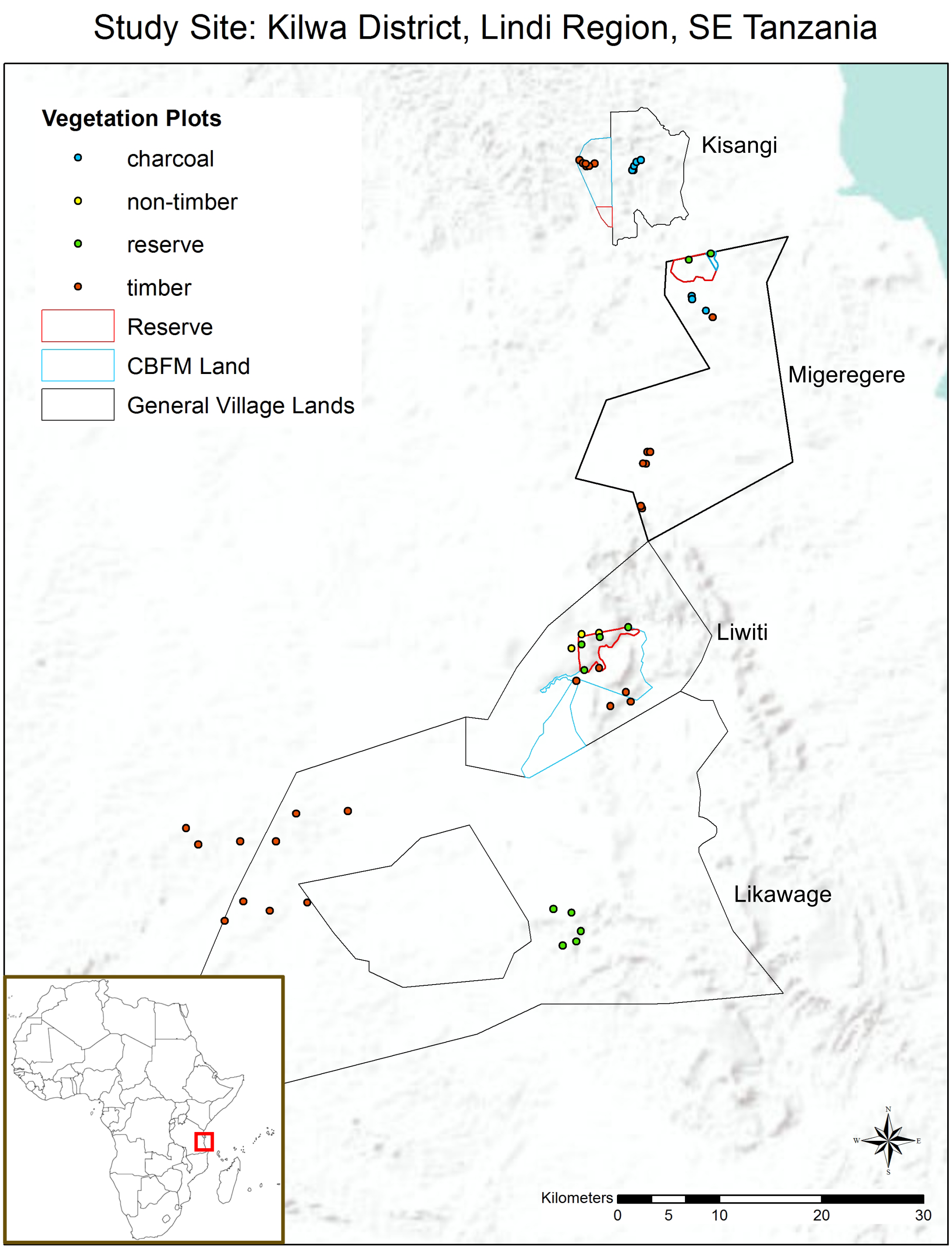
We surveyed four village forests in southeastern Tanzania with different combinations of management (community-governed forests with REDD+ projects and centrally-governed forests) and land uses (timber, charcoal, non-timber uses, and reserve). Since seedling recruitment is dependent on seed source, resource availability (light, water, nutrients), and disturbance regime, in order to compare effects of management and land use we sampled a wide set of environmental parameters to control for the natural variation across the landscape. By modeling seedling recruitment as a function of land management and use and environmental parameters, we are able to predict regenerative capacity of forests under various scenarios. This can then be used to guide decision-making on institutional structures for accomplishing REDD+ and other forest conservation goals. The particular questions we aim to answer are 1) How do biomass stocks and tree diversity compare between management types and among land uses? 2) Which edaphic, environmental, and anthropogenic factors most strongly effect seedling recruitment? 3) Which forestry management regimes and land uses have highest rates of seedling recruitment?

2. Methods

2.1 Study Site

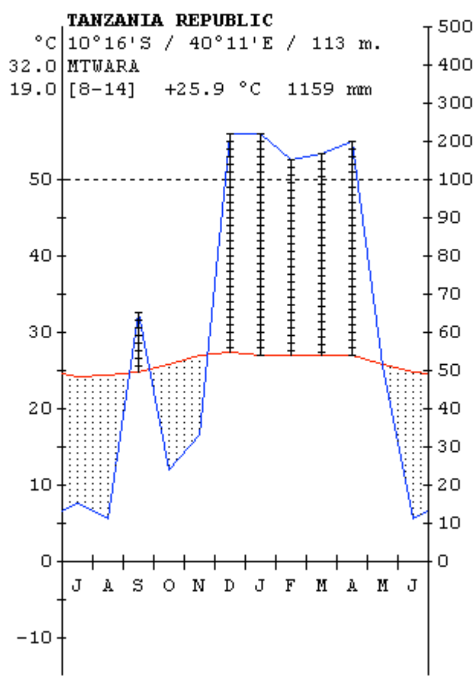
Our study site is located in Kilwa District of Southeastern Tanzania, 39°06’28” E, 8°39’32” to 9°33’57” S in the villages of Kisangi, Liwiti, Likawage, and Migeregere (see Fig. 1). The district receives a mean annual precipitation of 1000-1200 mm per year in two wet seasons, a brief rainy period in October, and a longer rainy period form December to March (see Fig. 2) (Hijmans *et al* 2005, WBCS 2009). The annual mean temperature of 25 °C is relatively steady throughout the year with slightly higher temperatures in the rainy season (~32 °C) (Hijmans *et al* 2005). Soils are typically sandy loams, well drained, slightly acidic, with low total exchangeable bases and cation exchange complex making them low in fertility (Frost, 1996).

Figure 1. Study sites in four villages were randomly stratified across land management and use areas, as well as abiotic gradients such as topography (shading on map represents relief).*



*Plots outside of management boundaries in Likawage were a result in the inconsistencies between government supplied coordinates of the boundaries and the ones maintained by the village. In this case we chose to categorize the area reflecting the village's maintained boundaries.

Figure 2. The climate diagram for Mtwara, Tanzania shows the bimodal rainy season and steady annual temperatures (WBCS 2009).



The vegetation in this region is Zanzibar-Inhambane coastal forest, which is mixture of Zambezian wet miombo woodland with coastal forest fragments and scrub (Schipper & Burgess 2003, White 1983). Zambezian miombo woodland is a fire-driven system distinguished from other African woodlands by the dominance of species in the family Fabaceae, subfamily Caesalpinioideae, particularly the genera *Brachystegia, Julbernadia, Isoberlinia* (Frost 1996). It occurs across the coastal plain of Africa except in areas where it is interrupted by hills or plateaus near the coast. Here, elevation increases precipitation and open woodlands grade into closed-canopy coastal forests (Burgess 1998). Eastern African coastal forests are considered climate relics of the Miocene that have receded to higher elevations (from changing climate and/or anthropogenic disturbances) and provide refuge for over 1300 endemic plant species (Burgess 1998). Biomass in coastal forests is high enough to suppress fire and species tend to be fire intolerant.

The Kilwa District has some of the largest coastal forest patches and remains virtually unsurveyed (Prins & Clark 2007). These forests are projected to have the highest rates of endemic species in the coastal forest chain and are considered a global biodiversity hotspot (Burgess 1998, Myer *et al* 2000). This region has also undergone a history of heavy logging, and improvements in infrastructure are opening areas that have previously been inaccessible (Prins & Clark 2007, Millege *et al* 2007). Brooks *et al* (2002) estimated that of the 25 biodiversity hot spots this region is the most threatened with extinction from habitat loss. The previously undescribed forests of Namatimbili, Mitundembea, and Ngarama were included within these surveys (near and around the villages of Likawage and Liwiti).

2.2 Land Management and Land Use Data

The governance structure of forest management was used as a proxy for resource ownership pool and local autonomy in resource allocation (thus use intensity), while land use data was used to as a proxy for the type of disturbance and associated ecological effects. For each of the four villages, meetings were held with the General Village Assembly, Village Council, and where applicable the Village Natural Resource Council, in which information was collected on land management, allowed land uses, restrictions, and enforcement (Mshale, in prep). Within the four sampled villages, there are two main land management regimes: 1) centrally-governed land and 2) community-governed ‘Community-based Forest Management’

(CBFM) land. Interviews were conducted with members of the village while visiting vegetation plots to verify land use history when possible (Andresen, unpublished manuscript). Through these discussions, four main land uses were identified across the two management types: timber (pit sawing and commercial), charcoal manufacturing, non-timber products (forest foods, fuel wood, and building material collection), and reserve.

Centrally-managed forests are tracts of land that experience use by local villages, regional governments and businesses, and the central government, so that they have the largest user pool, the least local autonomy in resource allocation, lower rates enforcement of forest rules and regulations. Illegal harvest of trees is common, and these areas can experience high pressure from commercial export markets for high-value timber and gas exploration which are permitted through the central government (see Milledge *et al* 2007). Most areas are open access, though there are centrally-managed reserves within this district. Uses within the open access areas can be categorized selective timber, charcoal manufacturing, and non-timber forest products including building materials, fuel wood, and food (Table 1).

Community-based Forest Management (CBFM) is a subset of Tanzania's Participatory Forest Management program and reflects recent national strategies to decentralize forest governance. This institutional arrangement grants autonomy to the village in rule making, resource allocation, and enforcement. Most areas within CBFM regimes are designated for sustainably-certified commercial timber harvest with the Forest Stewardship Council (FSC), and ~15% of the land is set aside as conservation zone. Benefits are structured so that local people directly receive revenue from logging operations, unlike the centrally-managed lands where revenue generated goes to the central government (Ball & Harrison 2010). In the Kilwa District, CBFM is facilitated by the non-profit organization Mpingo Conservation and Development

Initiative (MCDI). Targeted harvest species include African blackwood *Dalbergia melanoxylon* (mpingo) and other precious hardwoods. These areas are also being co-designated for REDD+ carbon crediting, uniquely pairing sustainable timber harvest with goals of reducing deforestation rates in these areas (MCDI 2010). The conservation zones were categorized as reserve areas for our study; they differ from general land reserves in that they are often intentionally placed near rivers or other sensitive or highly productive areas.

Table 1. Distribution of plots across land management, land use, and village.

Village	Community-managed Forest		Central Government-managed Forest				Grand Total
	timber	reserve	timber	charcoal	non-timber	reserve	
Kisangi	6	0	0	5	0	0	**11**
Likawage	0	0	10	0	0	5	**15**
Liwiti	2	4	3	0	3	0	**12**
Migeregere	0	0	7	3	0	2	**12**
Grand Total	**8**	**4**	**20**	**8**	**3**	**7**	**50**

2.3 Vegetation Plots

From June through August 2011 forest biomass, tree species diversity, and recruiting seedlings were measured using nested rectangular plots randomly stratified across biotic and abiotic gradients including forest type (coastal forest, closed canopy miombo woodland, open canopy woodland), elevation, topographic position, and proximity to villages and major roads (*n*=50). Location in UTMs, elevation, and distance from access roads were measured using a GPS (Garmin GPSMap 60Csx). Percent hill slope, canopy cover and grass cover were recorded using visual estimates. The species and basal areas were determined for all trees greater than 10.0 cm diameter at breast height (dbh, 1.3 m) ("adults") in 50 x 20 meter plots, 3.1 to 10 cm dbh ("saplings") in 20 x 20 m nested plots, and trees 3 cm dbh and less ("seedlings") were measured in a 20 x 2 m transect positioned across the center of the 20 x 20 m plot. This was considered adequate to measure reproduction of neighborhood adults as many miombo species have small dispersal distances (Campbell 1996). Heights were measured using a laser range finder (Laser Technology TruePulse 200). Common tree species were identified in the field with the help of local botanists and available species lists (i.e. Makonda 2010). When field identification was not possible, voucher specimens were sent to the University of Dar es Salaam Herbarium for identification or confirmation of field identification.

Four soil pits were dug along the center of each 50 x 20 m plot. One sample was taken from each of four different depths, 0-5 cm, 5-10 cm, 10-20 cm, 20-30 cm, using a standardized cylinder for a total of 16 samples per plot. Soil carbon and texture were determined at the University of Dar es Salaam soils lab. Soil organic carbon was measured by grouping together samples of each stratum per plot, giving an average of soil percent carbon per strata using standard methods. For soil texture data, all strata of soil samples were mixed for each plot giving an average value. Soil texture was determined using the hydrometer method (Bouyoucos 1962).

2.4 GIS Data

Frequency of fires in vegetation plots were estimated from remote sensing data (NASA LP DAAC 2011a, 2011b). All years with medium to high confidence fires from MODIS active fire product (2000-2011) and burned area product (2002-2011) were compiled using ERDAS Imagine (vers. 10.0, Intergraph) and ArcGIS (vers. 10.0, ESRI). Because the two MODIS product tend to identify different fire types within miombo systems (Roy *et al* 2008), active fire records and burned area records were summed for each plot. Multiple fire occurrences within a plot in one year were discounted as being incomplete initial burns and only counted as one fire occurrence.

The distances of vegetation plots from access roads and the Nangurukuru market were considered as a possible proxies for accessibility of resources and marketability, thus intensity of resource extraction (particularly relevant for charcoal manufacturing). Distances were determined using the network analysis function in ArcGIS 10.0 (vers. 10.0, ESRI). Vegetation plot coordinates (UTMs) were used for location and network pathways followed hand-digitalized road maps.

Precipitation and temperature data for the plots was extracted from the WorldClim data set in ArcGIS (vers. 10.1, ESRI), but there were no large differences across study sites at the resolution of data available and thus was not included in the analysis (Hijmans *et al* 2005)

2.5 Exploratory Data Analysis

Descriptive statistics and exploratory analysis was carried out using the R statistical computing environment (vers. 2.15.2, R Core Development Team). Richness, diversity, species accumulation curves, and population structure were determined for each plot using the R package Vegan (vers. 2.0, Oksanen 2011). Richness data included all size classes. Diversity (*D*) was calculated using Simpson's Diversity Index,

$$D_1 = 1 - \sum_{i=1}^{S} p_i^2$$

where p_i is the proportion of species i, and S is the number of species. High diversity approaches a D value of 1. Species accumulation curves were generated using the random accumulation method with 100 permutations (Gotelli & Colwell 2001) to see if we reached the asymptote of species sampled.

Aboveground biomass (B, Mg ha^{-1}) was calculated using an allometric equation developed from multiple miombo woodland tree species in northern Tanzania (R^2=.97) (Mugasha & Chamshama 2002),

$$B = (\Sigma (b_0 * d^b{}_1)) * S_i / 1000$$

where d is dbh in cm of each tree and b_0 and b_1 are constants 0.0625 and 2.553, respectively, giving tree biomass in kg. S_i is the multiplier transforming subplot biomass into kg ha^{-1} for saplings (S_i = 25) and adults (S_i = 10), which was then scaled to Mg ha^{-1} by dividing by 1000. Height data collected within the vegetation plots did not provide a better model fit and was excluded.

Abiotic and biotic environmental predictor variables (fire frequency, elevation, slope, soil texture, soil carbon, biomass, percent canopy cover, percent grass cover) and anthropogenic variables (distance to access road, distance to market, land use, land management) were explored in order to understand their impact on our response variable, seedling density. Because of the large number of predictor variables, Pearson's correlation coefficient (r) was calculated to assess co-variation and seedling density response (listed in Table 2, Appendix). For variables that were correlated >60% the strongest univariate variable was chosen. Before modeling, variables were standardized around their mean (obs-mean/2*SD) in order to investigate their relative explanatory power for each response variable.

2.6 Seedling Recruitment Model

Relative effects of land management and use on seedling recruitment were estimated using a Bayesian approach to generalized linear models. Seedling density (stems m$^{2\ -1}$) was used as an estimate of the likelihood that seedlings are able to survive on the forest floor given the set of environmental predictors. Seedling density (SD) at each plot i was modeled using a normal distribution, the likelihood:

$$SD_i \sim Normal\ (\mu, \sigma^2)$$

And process model:

$$SD_i = \alpha_{management(i),\ use(i)} + \gamma_1 \cdot fire_i + \gamma_2 \cdot biomass_i + \gamma_3 \cdot clay_i + \omega_i$$

Several sub-models with different combinations of fixed effects were tested to account for changing environmental conditions between plots. A sample of sub-models tested are shown in Table 3. In the final model shown above $\alpha_{management(i),\ land\ use(i)}$ represents the effects of each management type and land use combination on seedling density and the explanatory variables included in the final model, soil clay content, stand biomass and fire frequency. Parameters were estimated from non-informative normal prior distributions, $\alpha_i, \gamma_i \sim Normal\ (0, 1000)$ and $1/\sigma^2 \sim Gamma\ (0.01, 0.01)$. To account for unmeasured landscape variation effecting seedling densities, a spatially explicit error term (ω_i) was estimated as a function of the distance between areas *i* and *j* (d_{ij}), the rate of decay of correlation with distance (ϕ), and smoothing factor κ (Spiegelhalter *et al* 2012),

$$\omega_i \sim Normal\ (\mu, \tau_w)$$

and

$$\mu = exp[-(\phi\ d_{ij})^{\kappa}],$$

where $\tau_w \sim gamma\ (0.001, 0.001)$, $\phi \sim uniform\ (0.002, 5)$, and $\kappa = 1$.

Posterior densities of the parameters were obtained using Gibbs sampling (Geman and Geman 1984) in WinBUGS (vers. 3.2.2, Spiegelhalter *et al* 2002). Three chains of MCMC simulations with different initial conditions were run for 100,000 iterations, thinning every 100 iterations. Pre-convergence iterations were discarded and the model with the lowest DIC was chosen.

Table 3. Sample of seedling density (*SD*) sub-models tested. The model with the lowest DIC was used.

	Sub-model	**DIC**
A.	$SD_i = \alpha_{management(i),\ use(i)} + \gamma \cdot elevation_i + \omega_i$	-17.49
B.	$SD_i = \alpha_{management(i),\ use(i)} + \gamma \cdot biomass_i + \omega_i$	-37.12

C.	$SD_i = \alpha_{management(i),\ use(i)} + \gamma \cdot soil\ carbon_i + \omega_i$	-43.85
D.	$SD_i = \alpha_{management(i),\ use(i)} + \gamma \cdot clay_i + \omega_i$	-73.45
E.	$SD_i = \alpha_{management(i),\ use(i)} + \gamma_1 \cdot clay_i + \gamma_2 \cdot biomass_i + \omega_i$	-46.69
F.	$SD_i = \alpha_{management(i),\ use(i)} + \gamma_1 \cdot fire_i + \gamma_2 \cdot biomass_i + \omega_i$	-67.10
G.	$SD_i = \alpha_{management(i),\ use(i)} + \gamma_1 \cdot fire_i + \gamma_2 \cdot biomass_i + \gamma_3 \cdot clay_i + \omega_i$	-513.90
H.	$SD_i = \alpha_{management(i),\ use(i)} + \gamma_1 \cdot elevation_i + \gamma_2 \cdot biomass_i + \gamma_3 \cdot fire_i + \omega_i$	56.08

Predicted seedling density for each land management and land use combination ($SDp_{management(i),\ use(i)}$) was generated from model parameter estimates, using mean values for environmental predictors at each management-use combination. We then compared predictions from each land management and land use combination with a control, an average of seedling densities in all reserve areas. For that we calculated the probability of reaching recruitment rates as high or higher than the average in the control. The effect of each land management and land use (ELMLU) was estimated by calculating the ratio between that probability and the probability of the control (0.5). Values of ELMLU < 1 indicate a reduction of recruitment rates under that management-use combination, values >1 indicate an increase over the control.

3. RESULTS

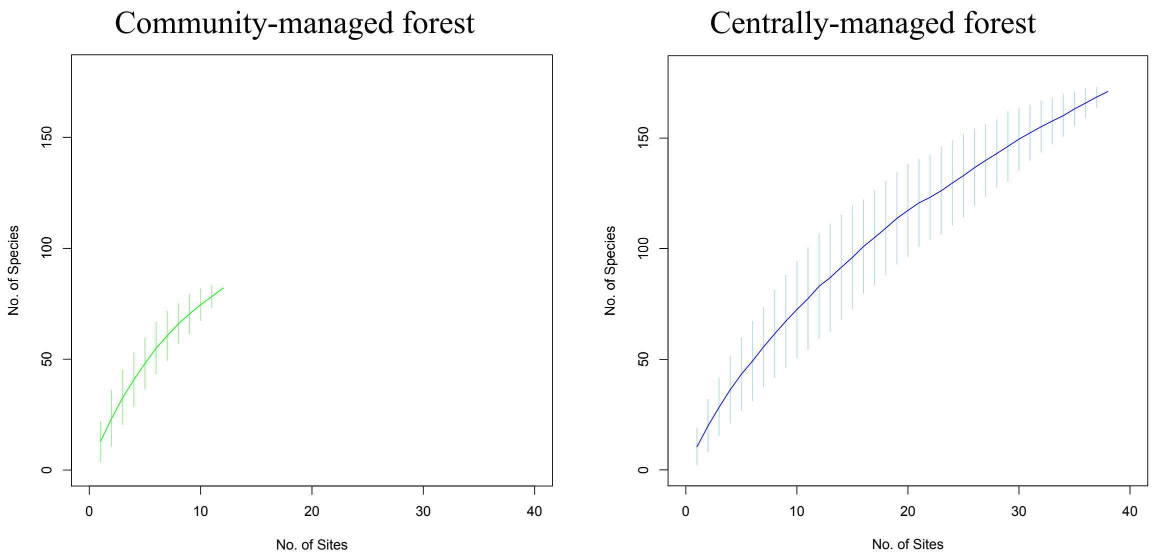
3.1 Forest composition and structure

Seventy-one percent of trees encountered (n = 2659) were identified to species level. Plots generally followed an inverse-J shaped population structure in which there are large numbers of seedlings and fewer adults. Thirty-four plots were located in miombo woodlands. These areas were dominated by *Julbernardia globiflora, Brachystegia tamarindoides, B. boehmii, B. speciformis, Markhamia obtusifolia, Millettia stuhlmannii, Pteoleopsis myrtifolia, Pterocarpus angloensis, Sterculia quinqueloba, Hymenocardia ulmoides, Spirostachys africana* in the overstory with a subcanopy dominated by *Combretum adenogonium, Diplorhynchus condylocarpon, Dalbergia melanoxyline,* and *Pseudolachnostylis maprouneifolia*. Sixteen plots were located within coastal forests. Composition was highly variable depending on forest fragment. The species comprising the highest basal areas are *Brachystegia tamarindoides, Bombax rhodognaphalon, Adansonia digitata, Maytenus undata, Hymnocardia ulmoides, Hymanaea verrocosa, Khaya anthotheca, Diospyros squarrosa, Carpodiptera africana, Grewia conocarpa,* and *Hugonia castanefolia. Afzelia quanzensis,* a UN Red List Threatened species was found in six plots, mostly the village of Liwiti.

3.2 Biotic and abiotic relationships

When comparing average tree species diversity (*D*) in each plot, it was slightly higher in community-managed forests (D = 0.84) but not significantly higher than centrally-managed forests (D = 0.82, p = 0.51). Centrally-managed non-timber use areas had the highest average diversity (D = 0.9), significantly higher than centrally-managed reserves (D =0.82, p =0.03) (Table 4). When comparing total richness within each management type (grouping plots together), richness was higher in centrally managed forests (n = 171 vs. 82); however neither species accumulation curve reached an asymptote and sample sizes are uneven (Fig. 3), and about 30% of individuals (n = 744) were not identified. Increases in both elevation and fire frequency decreased species richness. Richness and diversity were not correlated with biomass (r = -0.01 and 0.01 respectively; Table 2, Appendix).

Figure 3. The species accumulation curves (line = mean, bars = 95% confidence intervals) near, but do not reach, an asymptote.



Biomass is higher in community managed forests (138 ± 158 Mg ha^{-1}) but not significantly higher than general lands (58 ± 41 Mg ha^{-1}, $p = 0.11$) because of the high variance within management types. There were no significant differences in biomass between land use types across both management types, though there is a trend for community managed forests and reserves to have the highest means (Table 4). There were no clear relationships between biomass and predictor variables; biomass did not correlate with edaphic conditions or elevation as expected, nor were there strong relationships between biomass and distance from access roads and markets (Table 2, Appendix).

Table 4. Mean plot richness, diversity and biomass in land management and use combinations with standard deviation.

	Richness (*n*)	**Diversity (*D*)**	**Biomass (Mg ha^{-1})**
Central timber	13.35 ± 5	0.78 ± 0.13	47.13 ± 25
Central charcoal	15.62 ± 4	0.86 ± 0.05	53.51 ± 30
Central non-timber	13.33 ± 3	0.90 ± 0.02	77.39 ± 66
Central reserve	17.71 ± 5	0.84 ± 0.05	88.00 ± 65
Community timber	17.75 ± 3	0.87 ± 0.03	102.13 ± 79
Community reserve	17.25 ± 6	0.76 ± 0.10	208.87 ± 260
Central (all uses)	14.39 ± 5	0.82 ± 0.11	58.39 ± 41
Community (all uses)	17.58 ± 4	0.84 ± 0.08	137.71 ± 158

Active fire and burned area records only correlated by 0.20, justifying the choice to create a combined variable by summing both types of fire data. Areas that burned the most frequently are centrally-managed forests used for selective timber, as much as ten times over the ten-year monitoring period. The community-managed forests had much lower fire frequencies, with only one fire recorded over the ten-year monitoring period in four of the plots (Fig. 4, Appendix).

Seedling density was positively correlated with crown cover and biomass (r = 0.29, 0.44 respectively). There was also no indication that higher amounts of carbon in soils, which are associated with soil nutrients, facilitated seedling recruitment (r = 0.03). Soils with finer texture, i.e. high percentage of clay contents, did have a positive relationship to seedling recruitment (r = 0.16). Seedling density decreased with increasing fire frequency and higher elevations, similar to species richness (r = -0.23, -0.32 respectively).

3.3 Seedling Recruitment Model

The seedling recruitment model explained 83% of the variance in seedling densities (Fig. 5, Appendix). The highest seedling densities were found in community-managed forests, particularly reserves (1.12 ± 0.29 seedlings $m^{2\ -1}$), while the lowest densities were in the centrally-managed timber areas (0.44 ± 0.13 seedlings $m^{2\ -1}$). Seedling density was significantly higher in community-managed reserve areas (1.12 ± 0.29 seedlings $m^{2\ -1}$) than in centrally-managed reserve areas (0.53 ± 0.20 seedlings $m^{2\ -1}$). The environmental variables, fire, clay content, and biomass were not significantly different than zero in the model but were included because they provide more biologically realistic results and a lower DIC.

Table 5. Parameter estimates and mean posterior values with 95% credible intervals for each land management and land use combination (α). Letters (a, b) indicate significant differences.

	mean	SD	2.5 pc	97.5 pc
$\alpha_{\text{central timber}}$	0.44 a	0.1302	0.19	0.70
$\alpha_{\text{central charcoal}}$	0.77 a	0.1762	0.42	1.11
$\alpha_{\text{central non-timber}}$	0.50 a	0.2821	-0.03	1.09
$\alpha_{\text{central reserve}}$	0.53 a	0.1963	0.18	0.94
$\alpha_{\text{community timber}}$	0.82 ab	0.2076	0.44	1.26
$\alpha_{\text{community reserve}}$	1.12 b	0.2864	0.57	1.70
γ_1	-0.06	0.2002	-0.47	0.33
γ_2	8.67E-04	8.05E-04	-6.84E-04	0.002
γ_3	0.24	0.14	-0.05	0.53
ϕ	2.47	1.45	0.10	4.87
σ^2	87.84	250.50	3.75	726.30
σ^2_w	107.70	366.20	3.53	898.70

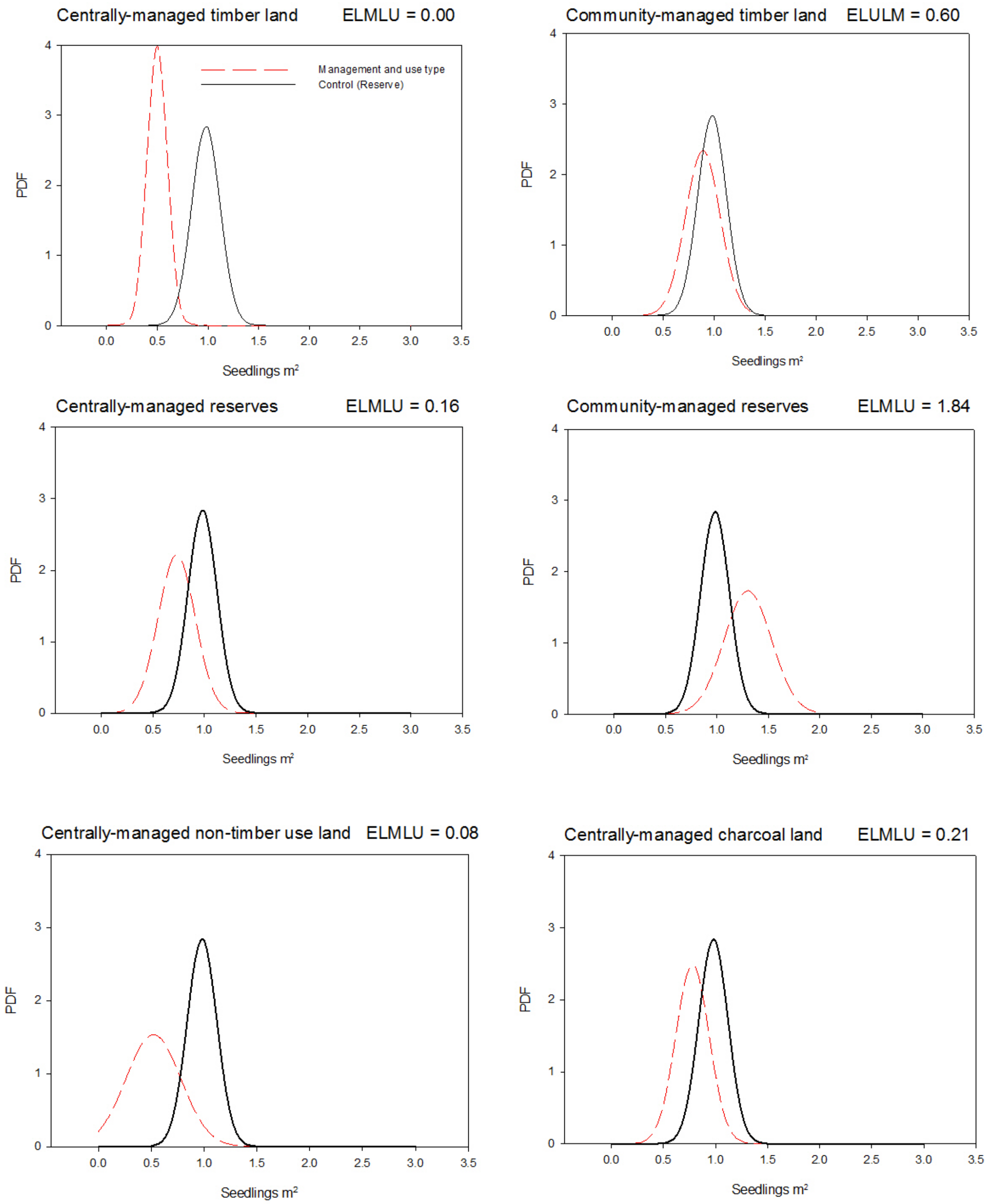
The probabilities (*P*) of seedlings within the treatments (land management and use combinations) reaching the mean seedling density of the control (mean reserve term, 0.98 ± 0.14 seedlings $m^{2\ -1}$) are displayed in Table 6, and the effect sizes of land management and land use (ELMLU) are displayed in Fig. 6. Community-managed reserve forests have a 92% chance of reaching mean values, and a positive effect size of 1.84. They are significantly higher in recruitment than all land uses in the centrally-managed forests. Community-managed timber areas have a 30% probability of reaching or exceeding the mean control seedling values and are significantly higher than centrally-managed timber forests. The centrally-managed use areas had low probabilities of reaching the mean control seedling numbers (0–11%) and no one land use was significantly higher than in recruitment than another (Table 6). Averaging land uses across management types, centrally-managed lands were significantly lower in recruitment than community-managed lands, with a 0% probability that recruitment will reach that of the community managed lands (Fig. 7).

Table 6. Predicted seedling values in each land management land use combination (*SDp*) with standard deviation and credible intervals, probability of reaching or exceeding densities of the control (*P*), and the effect size of the land management and land use combination on that probability (*ELMLU*). *ELMLU* value less than one indicates a decrease in probability of reaching the control while a value greater than one indicates an increase. Significant differences are indicated by notation (a, b, c).

	SDp	StdDev	2.5 pc	97.5 pc	*P*	*ELULM*
Central timber	0.50 a	0.10	0.30	0.71	0.00	0.00
Central charcoal	0.78 ab	0.16	0.46	1.10	0.11	0.21
Central non-timber	0.52 ab	0.26	0.01	1.05	0.04	0.08
Central reserve	0.73 ab	0.18	0.38	1.08	0.08	0.16
Community timber	0.89 bc	0.17	0.56	1.21	0.30	0.60
Community reserve	1.33 c	0.23	0.87	1.79	0.92	1.84
Central (all uses)	0.63 a	0.09	0.44	0.82	0.00*	0.00*
Community (all uses)	1.11 b	0.14	0.82	1.39	-	-

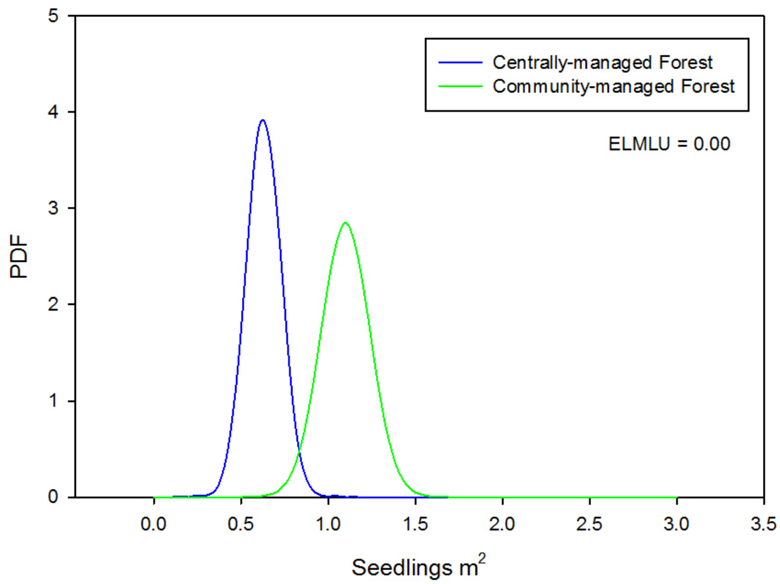
*Probability and effect size of centrally-managed forest as compared to community managed forest.

Figure 6. Probability distributions compare seedling densities within land management and use treatments (red dashed line) to a control (black line, reserve average). ELMLU (effect of land management land use) is the ratio between the probabilities of the seedlings densities reaching or exceeding the mean control densities.



Centrally-managed timber land
ELMLU = 0.00
Management and use type
Control (Reserve)
Community-managed timber land
ELULM = 0.60
Centrally-managed reserves
ELMLU = 0.16
Community-managed reserves
ELMLU = 1.84
Centrally-managed non-timber use land
ELMLU = 0.08
Centrally-managed charcoal land
ELMLU = 0.21
PDF
Seedlings m²
0.0
0.5
1.0
1.5
2.0
2.5
3.0
3.5
0
1
2
3
4

Figure 7. Probability distributions compare seedling densities of centrally-managed forests (blue line) to a community-managed forests (green line). ELMLU (effect of land management land use) is the ratio between the probabilities of the seedlings densities reaching or exceeding the community-managed land densities.



4. Discussion

The institutional structure of forest management and the associated land uses have been shown in many studies to affect forest biomass, stem density, deforestation rates, tree heights, and other indicators of forest health in miombo woodlands (i.e. Banda *et al* 2006, Blomely *et al* 2008, Chidumayo 2002, Luoga *et al* 2002). Although biomass and diversity showed no significant patterns between land uses and management structures, there was a trend towards community-managed forests being higher in both these qualities. Predicted seedling densities were significantly higher in community-managed forests, including within analogous land uses as the centrally-managed forests. This indicates that if goals of REDD+ and other deforestation reduction efforts are to succeed, they may function best within a community-managed governance structure.

Species accumulation curves did not reach an asymptote, therefore diversity and richness were not adequately sampled and comparisons are premature. Lower management-wide richness in community-managed forests could be a result of the smaller sample size, and not necessarily reflect the management type. Biomass estimates may also be difficult to compare between management types, as community-based forest management parcels are often designated based on the presence of harvestable timber species, confounding the effects of management on biomass. None of the biological parameters expected to be regulating biomass across the landscape (i.e. elevation, soil texture) showed strong relationships, making it difficult to make predictive models for regional biomass from site data. This could be from complex disturbance histories. Contrary to biomass, seedling densities represent short-term dynamics within a system. Short-term dynamics have been shown to not necessarily reflect long-term dynamics (Chesson and Huntly 1989), however the time frame of ecological responses to short-term disturbances is often most useful for conservation planning because projects often operate within these short time frames (Ezard *et al* 2010).

Seedling density was best predicted by above-ground biomass, fire, and the clay content of the soils. Though none of these parameters were significantly different than zero, they increased the fit of the model and are biologically relevant. The small size of the effect of the biomass parameter was a relict of it not being standardized and therefore cannot be directly compared to the effect size of fire and clay content. Biomass likely has a positive effect on recruitment because the system is dispersal or water limited rather than light limited, which is reasonable to expect in dry forests. The lower light levels also reduce competition from grasses which also promote fire (Campbell 1996). The positive effect of clay in the soils also suggests recruitment is water limited as clays have a high water holding capacity. This agrees with previous research that also concludes tropical forests are water limited (Zelazowski *et al* 2011).

Although many miombo species are thought to be fire tolerant or dependent on fire for regeneration (Campbell 1996), previous studies have shown that frequent (i.e. annual) and high intensity fires can result in rootstock mortality, no regeneration, and reduction of standing biomass (Ryan & Williams 2011, Chidumayo 2002, Luoga *et al* 2001). Fire patterns exist within a land use context through ignition sources, and more importantly, through changes in the vegetation which promote spread and connectivity (Archibald *et al* 2011). Fire not only creates top kill as a result of the heat but it is also associated with secondary effects such as reduced forest floor litter and soil carbon. Frequent disturbance can result in forest-floor dynamics changing to favor the establishment of grasses instead of tree seedlings, facilitating a transition to stable-state savanna (Hoffmann *et al* 2007). Areas that are low in soil fertility are particularly vulnerable to this transition (Hoffmann *et al* 2012). Since the natural rate of forest expansion is less than 15 m century^{-1} (Wiedemeier *et al* 2012), small but frequent perturbations can accumulate into net transitions towards lower biomass systems. Coastal forests may be particularly vulnerable to an increase in fire frequency as species tend to be less fire tolerant, resulting in a decrease in biodiversity.

Areas with the highest seedling densities were located in community-managed forests, both reserve and timber management areas. Probability density functions showed that timber, charcoal and non-timber lands under central management had low probabilities (0, 11, and 4% respectively) of reaching control levels of seedling densities. The low seedling densities on centrally-managed timber and reserve lands in comparison to the same land uses in community-managed areas indicates governance structure effects regeneration even within similar land uses. This could be explained by the intensities of disturbance experienced differently within different management regimes. Centrally-managed timber lands have a history of high rates of exploitation in this region; A 2007 publication reported that most high-quality timber being exported from Tanzania was from miombo woodland and coastal forest in southern Tanzania (~500,000 m^3 yr^{-1}), resulting in the removal of nearly all harvestable-size valued species from the centrally-managed land in the Kilwa District (Milledge *et al* 2007). Community-managed forests were not subject to this wave of exploitation, and Forest Stewardship Certified selective timber is in its early stages (Ball & Harrison 2010). If timber harvest becomes intensive, contributing to low biomass and increased fire connectivity, seedling recruitment could fall to low rates. For community-base forestry management operations to accomplish REDD+ goals, it will be crucial to maintain minimum levels of biomass and practice fire suppression or early, low-intensity burns, with special attention paid to drier and low fertility sites (also discussed in Barlow *et al* 2012).

Opportunity and implementation costs of REDD+ or other carbon sequestration projects in Tanzania from avoided agriculture or charcoal production have been calculated to ~US $15 Mg C $emissions^{-1}$ (Fisher *et al* 2011). When REDD+ is paired with timber harvest, incoming revenue is much higher than that offered purely by carbon markets (Ball & Harrison 2010) and projects are able to provide better alternative livelihood benefits, increasing participation. This comes at a compromise to reducing deforestation rates, and as this work shows it may also compromise future carbon storage capacity of forests if timber harvest is not carefully implemented.

5. Conclusion

Community-based forest management areas may serve a useful surrogate programs for REDD+ and other regional scale deforestation reduction efforts because it encourages responsible resource use though local control on resource allocation, rule-making autonomy, and direct benefits to the community. These institutional settings transfer to ecological outcomes by lessening intensity of resource extraction and the magnitude of disturbances experienced by a forest, and therefore we saw higher rates of regeneration in community-managed forests. Since forest regeneration is critical in maintaining regional biomass and carbon sequestration, community-based forest management will help meet REDD+ goals of reducing net forest loss. However, because it is also paired with timber harvest operations, careful attention will be necessary to retain threshold levels of biomass necessary to reduce fire connectivity, and provide seeds sources and safe seed sites.

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7. Appendix

Table 2. Pearson's correlation coefficients (*r*).

	elevation	slope	crown cover	ground cover	burned area	active fire	combined fire	%C 5cm	%C 10cm	%C 20cm	%C 30cm	Avg. %C	basal area	D	seedling density	richness	%sand	%silt	%clay	biomass
elevation	1.00	-0.24	-0.29	-0.08	0.50	-0.24	0.26	-0.14	-0.34	-0.06	0.07	-0.18	-0.14	-0.14	-0.32	-0.37	0.09	0.02	-0.18	-0.12
slope	-0.24	1.00	0.17	-0.02	-0.20	0.09	-0.10	-0.10	0.16	-0.05	0.26	0.07	0.29	0.01	0.22	0.01	-0.03	0.16	-0.06	0.30
crown cover	-0.29	0.17	1.00	-0.35	-0.39	-0.47	-0.54	0.11	0.18	0.17	-0.08	0.15	0.68	-0.05	0.29	0.42	-0.05	0.01	0.08	0.59
ground cover	-0.08	-0.02	-0.35	1.00	-0.06	0.22	0.07	-0.06	-0.17	0.09	-0.06	-0.07	-0.27	0.21	-0.05	-0.10	0.12	0.26	-0.23	-0.21
burned area	0.50	-0.20	-0.39	-0.06	1.00	0.19	0.86	-0.05	-0.06	-0.10	0.11	-0.04	-0.26	-0.37	-0.21	-0.45	-0.45	0.10	0.29	-0.25
active fire	-0.24	0.09	-0.47	0.22	0.19	1.00	0.67	-0.17	-0.01	-0.20	-0.20	-0.26	-0.31	-0.02	-0.12	-0.29	-0.18	0.14	0.13	-0.26
combined fire	0.26	-0.10	-0.54	0.07	0.86	0.67	1.00	-0.13	-0.05	-0.18	-0.02	-0.17	-0.36	-0.29	-0.23	-0.49	-0.43	0.15	0.29	-0.32
%C 5cm	-0.14	-0.10	0.11	-0.06	-0.05	-0.17	-0.13	1.00	0.28	0.39	0.02	0.81	-0.04	-0.46	0.03	0.01	-0.11	0.25	0.10	-0.06
%C 10cm	-0.34	0.16	0.18	-0.17	-0.06	-0.01	-0.05	0.28	1.00	-0.02	0.01	0.47	0.14	-0.17	0.14	0.14	-0.16	0.10	0.08	0.11
%C 20m	-0.06	-0.05	0.17	0.09	-0.10	-0.20	-0.18	0.39	-0.02	1.00	0.08	0.61	0.12	-0.43	0.09	0.02	0.17	0.00	-0.11	0.09
%C 30cm	0.07	0.26	-0.08	-0.06	0.11	-0.20	-0.02	0.02	0.01	0.08	1.00	0.44	0.01	-0.01	-0.16	0.06	0.09	-0.28	-0.05	-0.03
%C mean	-0.18	0.07	0.15	-0.07	-0.04	-0.26	-0.17	0.81	0.47	0.61	0.44	1.00	0.07	-0.48	0.03	0.08	-0.01	0.06	0.02	0.02
basal area	-0.14	0.29	0.68	-0.27	-0.26	-0.31	-0.36	-0.04	0.14	0.12	0.01	0.07	1.00	-0.03	0.37	0.12	-0.05	-0.02	0.04	0.97
D	-0.14	0.01	-0.05	0.21	-0.37	-0.02	-0.29	-0.46	-0.17	-0.43	-0.01	-0.48	-0.03	1.00	0.10	0.33	0.09	-0.30	0.15	-0.01
seedling density	-0.32	0.22	0.29	-0.05	-0.21	-0.12	-0.23	0.03	0.14	0.09	-0.16	0.03	0.37	0.10	1.00	0.32	-0.18	-0.05	0.16	0.44
richness	-0.37	0.01	0.42	-0.10	-0.45	-0.29	-0.49	0.01	0.14	0.02	0.06	0.08	0.12	0.33	0.32	1.00	0.04	-0.36	0.23	0.01
%sand	0.09	-0.03	-0.05	0.12	-0.45	-0.18	-0.43	-0.11	-0.16	0.17	0.09	-0.01	-0.05	0.09	-0.18	0.04	1.00	-0.38	-0.74	-0.05
%silt	0.02	0.16	0.01	0.26	0.10	0.14	0.15	0.25	0.10	0.00	-0.28	0.06	-0.02	-0.30	-0.05	-0.36	-0.38	1.00	-0.11	0.04
%clay	-0.18	-0.06	0.08	-0.23	0.29	0.13	0.29	0.10	0.08	-0.11	-0.05	0.02	0.04	0.15	0.16	0.23	-0.74	-0.11	1.00	0.01
biomass	-0.12	0.30	0.59	-0.21	-0.25	-0.26	-0.32	-0.06	0.11	0.09	-0.03	0.02	0.97	-0.01	0.44	0.01	-0.05	0.04	0.01	1.00

Figure 4. Fire frequencies estimated from MODIS burned area and active fire products. Dots represent vegetation plot location.

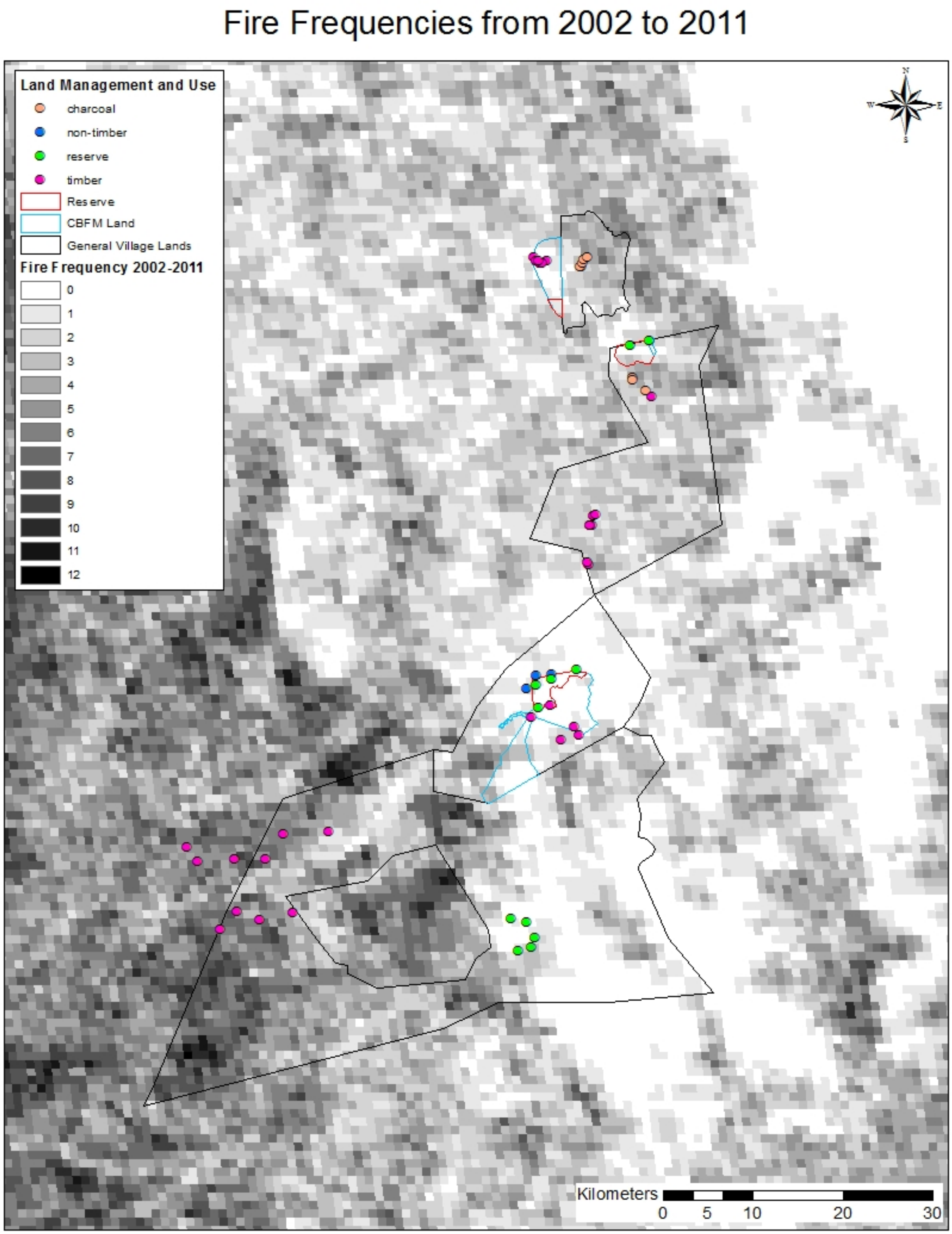

Figure 5. Seedling recruitment model predicted vs. observed values with 1:1 line.