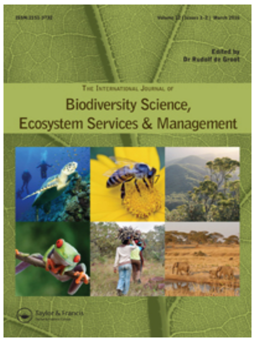

International Journal of Biodiversity Science, Ecosystem Services & Management

ISSN: 2151-3732 (Print) 2151-3740 (Online) Journal homepage: https://www.tandfonline.com/loi/tbsm21

Impacts of participatory forest management on species composition and forest structure in Ethiopia

Aklilu Ameha, Henrik Meilby & Gudina Legese Feyisa

INTERNATIONAL JOURNAL OF BIODIVERSITY SCIENCE, ECOSYSTEM SERVICES & MANAGEMENT, 2016
VOL. 12, NOS. 1–2, 139–153
http://dx.doi.org/10.1080/21513732.2015.1112305

Impacts of participatory forest management on species composition and forest structure in Ethiopia

Aklilu Ameha, Henrik Meilby and Gudina Legese Feyisa

Department of Food and Resource Economics, University of Copenhagen, Frederiksberg C., Denmark

ABSTRACT

The present study assesses the impacts of decentralized forest management on forest conditions in Ethiopian Montane forests. We compared observed densities of different tree species and size categories in forests managed by local forest user groups (FUGs) and the government. We used forest inventory data from 23,046 ha of contiguous forest managed by 74 individual FUGs. Topographical variables, including altitude, slope and aspect, were retrieved from Digital Elevation Model data for each FUG polygon. Generalized additive models and matching models were employed to analyse the effects of management and eliminate confounding factors. Findings show that altitude and slope were the topographical variables that had the strongest influence on species distribution. The overall densities of mature trees ha^{-1} and four individual species (*Afrocarpus falcatus, Schefflera abyssinica, Hypericum lanceolatum* and *Rapanea melanophloeos*) were higher in forests under participatory management (*p* < *0.01*). The three major commercial timber species *Juniperus excelsa, Afrocarpus falcatus and Hagenia abyssinica* constituted 49% of the total relative density and 39% the total relative frequency. In spite of the fact that inventories were carried out only 3–5 years after the forests had been handed over to FUGs, the observed patterns in vegetation density indicate that participatory management was more successful than government management in making forestry sustainable.

ARTICLE HISTORY
Received 30 January 2015
Accepted 14 October 2015

EDITED BY
Carsten Smith-Hall

KEYWORDS
Forest user group; generalized additive models; human disturbance; impact evaluation; propensity score matching; confounding variables; decentralized forest management

1. Introduction

Decentralized forest management has been claimed to hold considerable potential for halting tropical deforestation by ensuring tenure security and more responsible forest governance (Angelsen & Wunder 2003; Sunderlin et al. 2005; Poffenberger 2006; Ribot et al. 2006). This management approach comprises a range of efforts to involve people who live in and around forests, organized in forest user groups (FUGs), in forest management decisions. Different forms of decentralized forest management exist in different countries and are described using terms such as co-management, community-based forest management, community forestry, joint forest management and participatory forest management (PFM) (Lawrence 2007; Bowler & Pullin 2010).

Findings on the impacts of decentralized management on forest stocking have been ambiguous, but most studies have shown a decline in rate of deforestation and forest degradation after the introduction of decentralized forest management (e.g. Gautam et al. 2004; Somanathan et al. 2009; Coleman & Fleischman 2012; Takahashi & Todo 2012), but see also Oyono et al. (2012). A meta-analysis of 40 protected areas and 33 community-managed forests (Porter-Bolland et al. 2012) showed that the latter had a lower average rate of deforestation. Coleman and Fleischman (2012) analysed the impact of decentralized forest management on perceived forest conditions in case studies in four countries and, controlling statistically for confounding variables, they observed greater forest cover in community-managed areas in most of the cases researched. However, the study used perception-based assessment of forest conditions, and the results can be influenced by many factors that were not considered in the analysis (Lund et al. 2009).

The growing literature on impact evaluation emphasizes that impacts should be assessed for the degree to which changes in outcomes can be attributed to a particular intervention rather than to other factors. Such confounding factors can to some extent be controlled through evaluation designs, such as quasi-experimental designs (Ferraro 2009). The absence of consistent and rigorous environmental impact evaluation studies has been mentioned as a major hindrance to choose from available forest governance regimes (Ferraro & Pattanayak 2006; Caplow et al. 2011). The number of studies that employ quasi-experimental designs to evaluate impacts of conservation policies has grown in recent years (Caplow et al. 2011), but the majority have aimed to assess the impacts of protected areas (e.g. Andam et al. 2010; Ferraro et al. 2011; Pfaff et al. 2014) and

CONTACT Aklilu Ameha Aklilu.yt@gmail.com

few have focused on the impacts of community-based forest management (Somanathan et al. 2009). Particularly, the scarcity of impact studies is obvious in African conditions (Porter-Bolland et al. 2012). Besides, according to Bowler and Pullin (2010), only 10 out of 42 studies analysing the impact of decentralized forest management investigated factors that may have confounded the direct comparison of forests under decentralized management with forests under other management regimes.

Decentralized forest management, such as PFM, is meant to restrict the access to resources that were previously considered open to all. Unfortunately, increased regulation of a specific forest may result in aggravated rates of degradation in adjacent forests, especially where these continue to be the responsibility of government authorities. Such undesirable consequences, a 'leakage effect' of PFM, are reported in several studies (Chakraborty 2001; Schreckenberg & Luttrell 2009; Bowler & Pullin 2010). Takahashi and Todo (2012), for instance, found that in Ethiopian forest areas under PFM, the forest cover increased by 1.5% in the 2 years following the establishment of FUGs; during the same period, the forest cover of adjacent forests under government management decreased by 3%. Gobeze et al. (2009) showed that seedling and sapling densities increased in PFM forests compared to the adjacent non-PFM forest in Bonga, Ethiopia. None of these studies, however, considered whether the initial conditions of the forests under the two management regimes were comparable.

Inspired by the absence of rigorous empirical studies on the impacts of decentralized forest management, particularly in Africa, the present study was carried out in Ethiopia where the forest cover is believed to have declined from about 40% at the end of the nineteenth century (Dessie & Christiansson 2008) to 4% in recent years (Earth Trends 2007). PFM pilot projects were initiated in the 1990s to halt the deforestation, and since 2010, a national program to scale up PFM is being implemented, apparently without making use of the experiences gained from the pilot projects (Limenih & Temesgen 2011). Today about 667,498 ha of forest land is under management of 556 FUGs and 123 FUG cooperatives in Ethiopia (Ameha et al. 2014a). Our study site is located in the south-eastern highlands of Ethiopia, in Dodola district, where the first PFM pilot project was established in 2000. Forests in Dodola thus offer the longest experience with PFM in Ethiopia.

The purposes of the study were (1) to analyse differences on the densities of various tree species and size categories between forest under participatory management and forest under government management in Dodola District, (2) to identify the effects of confounding topographical and structural variables on the distribution and density of different tree species in the two case forests and (3) to analyse the stand structure and species composition in the forest. The results of the study provide empirical evidence that can inform future PFM policies in Ethiopia and elsewhere.

2. Site description and methods

2.1. Study site

The Adaba-Dodola forest area was chosen for this study for two reasons: (1) it is the oldest PFM area in Ethiopia, and (2) extensive forest mensuration has been carried out in the area, and spatial data are available both for forest areas already under PFM and for those in transition from government to participatory management. These types of data are not available from any other PFM area in Ethiopia.

The Adaba-Dodola forest is located on the northern slopes of the Bale Mountains in the south-eastern highlands of Ethiopia (Figure 1). The Bale Mountains form the largest continuous area in Africa with elevations exceeding 3000 m, and they therefore constitute the most extensive area of alpine and subalpine Ericaceous vegetation on the African continent (Uhlig & Uhlig 1991).

The Adaba-Dodola forest is bordered by vast agricultural plains (more than 2000 km^2), located at altitudes of about 2400 m and surrounded by mountain ranges with peaks that reach over 3700 m (Schmitt 2003; Johansson et al. 2012). The livelihoods of people living in the Dodola area are mainly based on agriculture, forestry and livestock production (Ameha et al. 2014b). The main forest products harvested are fuelwood (including for charcoal production), grass for livestock, and roofing, tree fodder, timber and construction wood. The forest has been inhabited by people for centuries. Anthropogenic disturbances such as expansion of agricultural land, browsing or grazing of livestock, extraction of wood and wildfire have progressively reduced the size of the forest from 140,000 ha in the early 1980s to 53,000 ha in 1997 (Kubsa et al. 2003). The annual deforestation rate between 1993 and 1997 was 3% (Tadesse 1999).

2.2. PFM approach

In the PFM arrangements of Ethiopia, the government holds ownership of the forest, while the local communities, organized in FUGs, have use rights. The first FUGs, named '*wajib*' in the local language, were established in 2000 in Dodola district with support of the German Government (GIZ) (Kubsa et al. 2003). According to the Dodola district forest

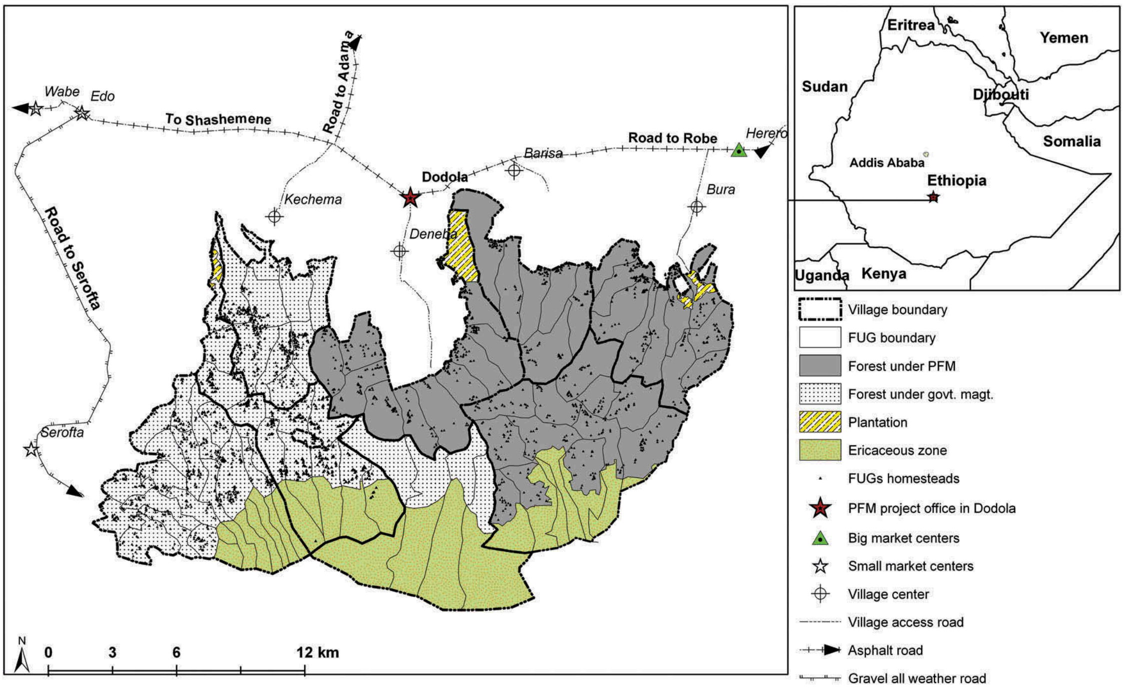

Figure 1. Location of the study site, boundaries of forest user groups and forest under participatory and government management in the Dodola forest.

enterprise office (the local forest authority), 101 FUGs with 2786 member households were established during 2000–2010 in 10 villages. These user groups manage 40,576 ha of forest, including agricultural plots and settlements located within the forest boundary, 8649 ha of *Erica* moorland and 1067 ha of plantation forest (*Cupressus lusitanica* and *Eucalyptus* sp.).

The main criterion for FUG membership is living inside the designated PFM forest boundary. The forest area in a given village was divided into blocks with an average area of 360 ha (Figure 1), and user groups consisting of a maximum of 30 households were formed in each block. FUG members were granted exclusive forest product use rights, including grazing and cultivation of agricultural plots inside the forest that they had been using before the FUG establishment. In return, FUG members are obliged to fulfil the following: (1) keep or improve the forest cover, (2) prevent establishment of new homesteads and expansion of farmland, (3) restrict use by non-members and (4) pay an annual forest rent to the government, US\$ 1 ha^{-1} $year^{-1}$ for tree-less areas in the forest to encourage user groups to increase forest cover of their designated area (Kubsa et al. 2003; Tesfaye et al. 2011).

2.3. Vegetation, soil and climate

The vegetation in the study area mainly belongs to the afromontane and afroalpine categories (Friis 1992), with distinct altitudinal zonation (Stipl 1988; Fichtmüller 1997). The soils are generally silt or clay soils, which have a depth of more than 1 m on gentle slopes and in valleys, but which are shallow on steep slopes and at the tops of ridges (Asrat et al. 1997). Soil organic carbon stocks vary with vegetation type and with differences in microclimatic conditions induced by topographical factors (Yimer et al. 2006).

The precipitation is bi-modal with a main rainy season from July to September and a short rainy season in March and April. The minimum and maximum temperatures recorded in the low- and mid-altitude range (2450–2600 m above sea level) were 10 and 23°C, respectively. Average rainfall in this altitude range was 850 mm. At high altitudes, about 3000 m, the minimum temperature was 5°C, the maximum was 23°C and the average annual rainfall was 1280 mm (Schmitt 2003). Around 3400 m, the annual rainfall was 1740 mm, and the average maximum and minimum temperatures were 15°C and 5°C during the rainy season and 22°C and 2°C during the dry season (Johansson et al. 2012).

2.4. Data

2.4.1. Forest inventory data

We used forest inventory data collected between October 2004 and December 2005 from 23,046 ha forest which are managed by 74 FUGs that were established in two separate phases. In the first phase (2000–2003), the management of 13,375 ha forest was transferred to 36 FUGs in four pilot PFM villages (Adele, Bura, Danaba and Barisa). In the second phase (2005–2006), 9671 ha forest were transferred to 38 FUGs in three adjacent villages (Qachema, Harageneta and Alentu). Thus, in 2004–2005, when the inventory was conducted, 13,375 ha forest were under participatory management and 9671 ha were still under government management. We use the term 'forest under participatory management' for FUGs established in the first phase (presented as dark grey areas in Figure 1) and 'forest under government management' for second-phase FUGs established shortly after the inventory (presented as the light grey areas in Figure 1).

The quantitative inventory data describing the forest composition were collected by Dodola District Forest Enterprise Office. A detailed account of the inventory sampling design is provided by Amente et al. (2006). Plots were circular with an area of 2000 m^2 and were located at the intersection points of a 100 m × 100 m square grid. The total number of plots assessed in the 74 FUGs was 20,655; the sampling intensity was 18%. In each plot, data were collected on potential crop trees (PCTs) and mature trees (MTs). PCTs are trees of economically desirable species that are vigorous and have the potential to develop a high timber quality (Lamprecht 1989). The criteria used to select PCTs in the study area were (1) healthy young trees with undamaged trunks, (2) trees taller than 2 m, assuming that trees this tall will escape further browsing damage and (3) commercial timber species with a diameter less than 40 cm and non-timber species with a diameter less than 25 cm diameter at breast height (dbh). Trees with diameters exceeding these limits were considered as MTs.

The different tree species occurring in the forests were further classified locally into two groups based on the size they reach at maturity. All commercial timber species attaining diameters larger than or equal to 40 cm at maturity were categorized as group one (GI) and species that achieve diameters larger than or equal to 25 cm at maturity were categorized as group two (GII). The species included in GI were *Juniperus excelsa* M. Bieb., *Afrocarpus falcatus* (Thunb.) C.N. Page, *Hagenia abyssinica* J.F. Gmel., *Ekebergia capensis* Sparrm., *Schefflera abyssinica* (Hochst. ex A. Rich.) Harms, *Olea europaea subsp. africana* (Mill.) P.S. Green and *Maytenus spp*. Species included in GII were *Pittosporum viridiflorum* Sims, *Hypericum lanceolatum* Lam., *Rapanea melanophloeos* (L.) Mez, *Erica arborea* L. and a residual group termed 'Others', which included, e.g., *Nuxia congesta* R. Br. ex Fresen. and *Buddleja polystachya* Fresen.

Within each plot, the number of stems and dbh of trees were recorded for GI species in three size classes: ≤20 cm; 20–40 cm and ≥40 cm and for GII species in two size classes: ≤25 cm and >25 cm. When dense clusters of young trees were encountered, a minimum distance of 4 m between selected PCTs was applied to reflect their future need for growing space, assuming that other individuals that were not selected would be removed.

2.4.2. Topographic models

The study area is characterized by mountainous terrain, and the effects of PFM establishment may be confounded with topographical factors. We therefore examined the influence of topographic variables on the density of each of the tree categories. A range of topographical variables, including altitude, slope and aspect, were estimated for each FUG polygon using the Global Digital Elevation Model, which is a product of the Ministry of Economy, Trade and Industry of Japan and the National Aeronautics and Space Administration. The data set was obtained from the web page of the United States Geological Survey Earth Explorer (USGS 2013). Altitude, slope and aspect variables were estimated using ArcGIS version 10.1 (Figure 2).

2.5. Data analysis

2.5.1. Generalized additive models

A generalized additive modelling (GAM) technique was used to construct regression models expressing the effect of management while also controlling for (possibly confounding) effects of topography and spatial location. GAM models were chosen because this model type allowed us to simultaneously model the effect of altitude using a spline function and estimate linear parameters expressing the effects of several other variables (for details see Guisan et al. 2002). We also used GAM models to select variables for the matching models described below. The models were prepared using the GAM procedure of the SAS (v. 9.2) software package.

The response variables in our data set were densities of individual species, PCTs and MTs ha^{-1}. Due to right-skewed density distributions of most species and the presence of many zero values for some species, mainly *A. falcatus, O.europaea, E.capensis* and *P.viridiflorum* – partly as result of their limited ecological range and partly because of human disturbance – we used square root-transformed density values to obtain residuals that behave randomly. For these data, we applied a Gaussian error distribution with an identity link function in the GAM. The models were reduced step by step by removing the least significant variable, one at a time, until no more variables could be dropped ($p < 0.1$). The predictor variables used in the final models were a dummy variable specifying the management regime, which is 1 for forest under participatory management and 0 for forest under government management; the mean altitude; the percentage of area with slope >15°; percentages of the area facing west, east and south; area of the FUG (in ha); area of Ericaceous shrub (in ha) and walking distance to the nearest market (in hours). Predictor variables such as the number of FUG members, population size and forest area per member were omitted from the model as they were not statistically significant for any of the response variables.

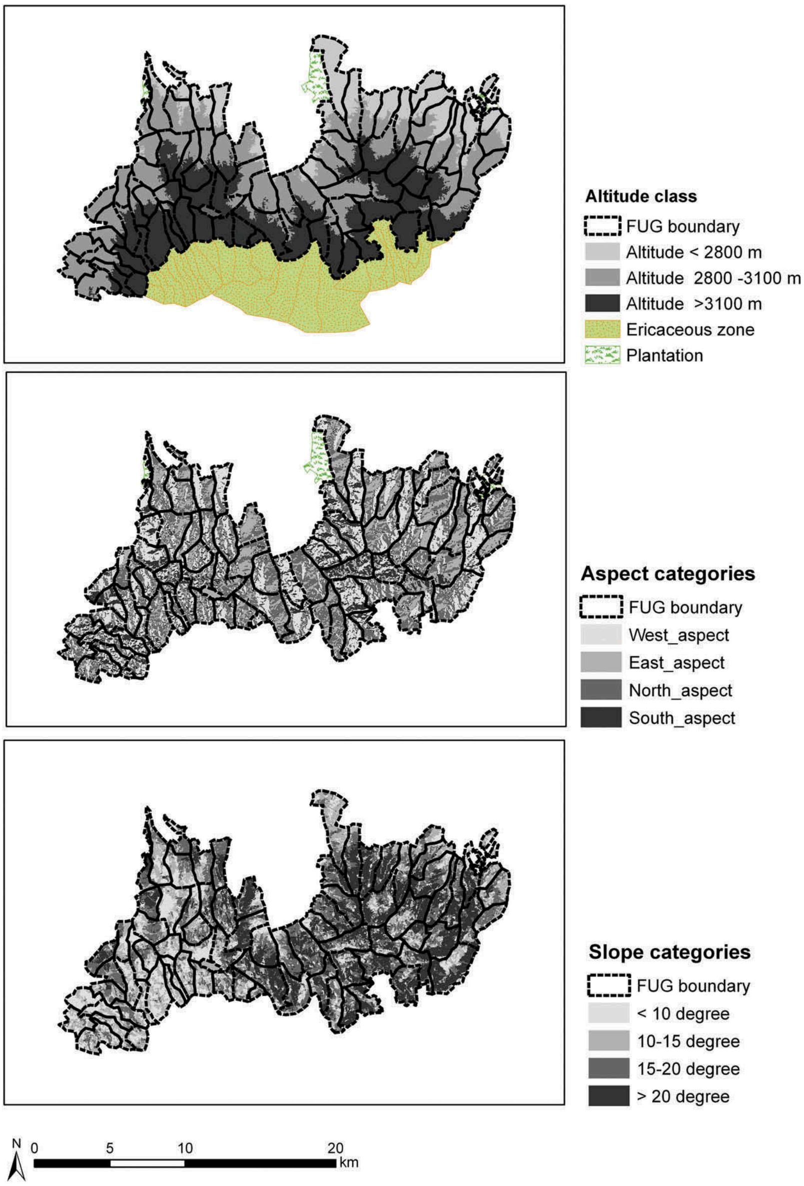

Figure 2. Topographic characteristics of the individual FUG areas (altitude, aspect and slope categories).

2.5.2. Matching models

The villages where PFM was implemented in the first phase were not chosen randomly, and hence there was selection bias. Discussion with key informants and review of project documents showed that the four pilot villages were selected neither on the basis of severe threats to the forest nor on the basis of high conservation potential. Rather, it appears that selection was mainly based on ease of access to pilot villages from Dodola town, where the offices of the forest department and the PFM pilot project were situated, and to the main highway (Figure 1). Non-random selection of PFM forest implies that there might be observable factors that differentiate the forest areas under PFM from those under government management. If this is not corrected for, it may lead to inaccurate conclusions on the difference in densities of trees under the two management regimes.

The FUGs varied with regard to topographical and structural variables, and the effects of these on density of different species and size categories may be confounded with the management regime. To capture the effects of the covariates, FUGs under participatory management (treatment group) were matched with areas under government management (control group) with similar propensity scores (PS). The applied unit of analysis was the area (a polygon) of forest assigned to each user group. Calculating PS reduces the need to match FUGs on every single relevant explanatory variable. If there is no selection bias conditional on the n-dimensional vector of explanatory variables, then there is no selection bias conditional on the 1-dimensional PS (Rosenbaum & Rubin 1983).

We selected variables for matching the two management regimes that were assumed to affect species distribution and density and that were observed to be significant in the GAM models. Moreover, the variables chosen were either measured before the PFM program started (2000) or assumed to remain stable over time. For instance, we omitted variables on the number of members and the total population living within an FUG, forest area of the FUG (including size of Ericaceous heather land zone) which formed part of the basis for establishment of FUGs and delineation of FUG areas. Instead, we emphasized on topographical variables, which have also been pointed out by previous studies as important determinants of forest condition outcomes (Agrawal & Chhatre 2006). The variables selected for matching were walking distance to market in hours, altitude (percentage of area lower than 2800 m and percentage of area higher than 3100 m); percentage of area facing east or west and percentage of area with slopes less than or equal to 10° and above 15°.

Walking distance to market has direct influences on density and composition of species. FUGs close to market can easily transport forest products in bulk that negatively influence stocking in the forest. To the contrary, those farther away from market have difficulty of transporting, hence less pressure on the forest. In mountain regions, topography is a key determinant of conditions affecting forests, including microclimate, soil properties and disturbances (Brown 1994). Altitude determines temperature, moisture and CO_2 pressure and affects the type and density of individual species both negatively and positively. Altitude has an important influence on species composition and conditions in mountain regions as it affects a host of other variables, including temperature, accessibility and ease for agricultural activities. Hence, altitude influences the density of individual tree species both positively and negatively. Slope and aspect determine solar radiation received, terrain stability, erosion and moisture (Bader & Ruijten 2008) and, in our case, access to harvest of trees and possibility of converting the forest to cropland. We selected percentage of area facing east or west because Ethiopia is close to the equator radiation difference between surfaces that face north and south would be minimal. In the study area, east facing slopes receive stronger radiation in the morning to noon and heat up earlier than west facing. West facing heat up in the afternoon while the east begins cooling down. In the night, thermal contrast between west and east faces could lead to differences such as fog formation. In addition, prevailing winds are from east to west during certain times of the year, and east-facing slopes would receive more precipitation than west-facing slopes. This may imply that the duration of the growth season becomes longer on one side of a hill than on the other.

Previous studies have documented the impacts of slope and aspect on vegetation type and hydrology and on the composition of vegetation and the distribution of species (Ganuza & Almendros 2003). Topographic summary variables were therefore calculated for each of the 74 FUGs.

A probit model (Appendix 1) was applied to estimate PS using the control/matching variables mentioned above. We applied two propensity score matching methods: Epanechnikov kernel matching with a default band width of 0.06 and nearest neighbour matching with three neighbours and caliper 0.06 with bootstrapped standard error of 1000 replications (for details on matching models, see, Becker & Ichino 2002; Caliendo & Kopeinig 2008). Both matching models omit those FUGs from the treatment group whose PS turn out to be higher than the maximum or lower than the minimum score among observations from the control group. The outcome variables were differences between forest under participatory management and forest under government management on the densities of PCTs and MTs of GI and GII species and individual species.

To check whether the distributions of variables between treatment and control groups were balanced or not, we first employed a standardized bias and t-test for difference, as proposed by Rosenbaum (2002). Second, we ran a probit model as suggested by Sianesi (2004) using the sample after matching to compare the pseudo-R^2 with the corresponding value obtained from the probit model prepared before matching. Finally, a likelihood ratio test was performed to check whether all the coefficients of the after-matching probit model could be zero. The analysis was carried out using stata software version 11 and the psmatch2 programme developed by Leuven and Sianesi (2003).

2.5.3. Vegetation stand structure

The structure of a stand, i.e., the distribution of trees by species and size is the result of species growth habits, environmental conditions and management practices (Husch et al. 2003). From an ecological perspective, species composition at a particular spatial scale is commonly viewed as having three components: frequency, abundance and dominance.

Species composition in this study was described by relative density and frequency. Relative density of each species was calculated as the number of stems of the species per plot divided by the total number of stems (all species) per plot. The relative frequency was calculated as the frequency [0; 1] of the species divided by the sum of frequencies across all species. Both variables were expressed in per cent.

3. Results

Topographic characteristics of the FUGs in Dodola are illustrated by the maps in Figure 2. It appears that the shapes and sizes of most FUGs, relative to the scale of the terrain, imply that topographical variables vary between FUGs. The descriptive statistics (Table 1) reveal that the characteristics of forest areas under participatory management and under government management differ on many variables. For instance, looking at some of the topographic variables in Table 1, 27% of the forest area under participatory management is located at relatively low altitudes (2500–2800 m) and 65% is on slopes steeper than 15°. By contrast, only 9% of the forest area under government management is located at altitudes of 2500–2800 m, and 65% is located on slopes less than 15°. It also appears that the densities of PCTs, MTs and individual species were higher in forest under participatory management, and in most of cases, the difference is statistically significant at the 1% level. However, topographical variables influence species density and distribution strongly. Therefore, this crude comparison of mean differences in densities of PCTs, MTs and individual species between the two management regimes is inadequate as a basis for assessing differences between management regimes, and we therefore proceed with the two approaches (they have been presented in the methods section) that control for the confounding variables.

3.1. Influence of topographic variables

Table 2 shows parameter estimates for the GAMs (using a smoothing spline to model effects of altitude) of the effects of topographical and structural variables on square root-transformed densities of individual species, PCTs and MTs of GI and GII species. Altitude generally has the largest influence on species distribution and density, followed by slope and two of the aspect variables (west and east).

The coefficients on management, the variable that identifies the impacts of PFM on density of trees, shows that the densities of *A. falcatus, S. abyssinica, H. lanceolatum* and *R.melanophloeos*, were higher in forests under participatory management than in forest under government management, all other variables being con-

Table 1. Summary statistics for forest under participatory management and forest under government management: mean values with standard deviations in brackets, minimum and maximum.

Variables	Description	Particip. Management (n = 36) Mean	Min	Max	Gov't management. (n = 38) Mean	Min	max	Difference[a]
Structural	Distance to market (walking hr)	2.4 (0.8)	1.1	4.0	2.1 (0.8)	1.2	4.0	**
	Area of FUG (ha)	372 (84)	179	634	255 (108)	73	461	***
	Members (number)	26 (5)	11	30	27 (4)	13	30	NS
	Population size	173 (44)	64	970	181 (155)	63	270	**
Altitude (m): proportion of area (%)	2500–2800	27 (21)	0	100	9 (34)	0	86	***
	2800–3100	42 (37)	0	99	43 (28)	0	100	NS
	>3100–3400	31 (42)	0	100	48 (35)	0	103	NS
Slope: Proportion of area (%)	≤10 degrees	14 (16)	7	27	32 (5)	7	75	***
	>10 and ≤15 degrees	20 (11)	11	51	33 (10)	9	53	***
	>15 and ≤20 degrees	28 (15)	10	58	26 (10)	3	61	NS
	>20 degrees	37 (11)	4	59	8 (16)	0	40	***
Aspect: Proportion of area (%)	East facing	41 (17)	7	83	30 (23)	1	62	**
	West facing	39 (17)	5	79	51 (21)	20	91	**
	South facing	5 (3)	1	24	5 (5)	0	16	NS
	North facing	16 (5)	5	35	15 (6)	3	26	NS
	East west facing	80 (6)	11	38	81 (5)	8	28	NS
	North south facing	20 (5)	62	89	19 (6)	72	93	NS
Density of trees (stems/ha)	PCT GI	43 (24)	4	112	20 (29)	0	106	***
	PCT GII	34 (15)	4	90	17 (23)	0	67	***
	Total PCT	77 (32)	8	148	36 (29)	1	138	***
	MT GI	34 (12)	7	65	21 (15)	4	53	***
	MT GII	23 (13)	1	82	13 (24)	0	50	NS
	Total MT	57 (15)	18	98	34 (22)	5	67	***
	Total PCT and MT	134 (34)	83	238	71 (37)	18	147	***
Species density (stems/ha)	*A. falcatus*	11 (13)	0	36	1 (4)	0	21	***
	J. excelsa	49 (24)	4	99	29 (22)	0	82	***
	H. abyssinica	6 (4)	0	18	6 (4)	0	19	NS
	O. europaea	5 (6)	0	28	1 (3)	0	10	***
	E. capensis	1 (2)	0	10	0.1 (0.2)	0	1	**
	Maytenus spp.	4 (3)	0	9	2 (5)	0	27	***
	S. abyssinica	2 (2)	0	6	1 (1)	0	4	NS
	P. viridiflorum	1 (1)	0	4	0.2 (0.5)	0	3	***
	H. lanceolatum	18 (22)	0	78	7 (8)	0	33	NS
	R. melanophloeos	16 (12)	0	45	7 (6)	0	21	***
	E. arborea	13 (18)	0	66	8 (16)	0	59	*
	Others	11 (9)	0	32	6 (10)	0	42	***
	Total	135 (36)	83	238	71 (34)	19	147	***

Notes: [a]Significance levels: NS not significant, *$p < 0.1$, **$p < 0.05$, ***$p < 0.01$. As the distribution of the data was not in agreement with the assumption of a normal distribution, we used the Wilcoxon/Mann–Whitney test to test the difference between the two groups.

Table 2. Parameter estimates and significance tests[a] for Generalized Additive Models (GAM) used to estimate effects of topographical and geographical features on square root-transformed density of individual species and species groups (ha^{-1}).

Species (number/ha)	Constant	Magt. (1 = PFM)	Slope > 15°	Aspect West (%)	Aspect East (%)	Aspect South (%)	FUG Area (ha)	Erica Area (ha)	Market Distance(hr)	Mean Altitude	Spline (Mean altitude) [b]
A. falcatus	21.58***	1.38***	NS	−0.01*	NS	−0.06*	−0.003**	NS	NS	−0.01***	***
J. excelsa	23.83***	NS	0.04***	−0.01*	NS	NS	NS	−0.004***	NS	−0.01***	NS
H. abyssinica	−3.05	NS	NS	−0.05***	−0.05***	NS	0.003***	NS	−0.33***	0.003***	***
O. europaea	9.94***	NS	0.02***	0.03*	0.03**	NS	NS	NS	−0.26**	−0.004***	***
E. capensis	5.02***	NS	NS	NS	−0.004*	NS	NS	NS	NS	−0.002***	*
Maytenus spp.	4.52***	NS	0.02***	NS	NS	NS	0.002**	NS	−0.39***	−0.001**	***
S. abyssinica	−4.89***	0.32***	NS	0.01**	NS	0.03**	0.001**	NS	NS	0.002***	***
P. viridiflorum	−0.96	NS	0.01***	NS	NS	NS	0.00**	0.001*	NS	0.000	NS
H. lanceolatum	−13.60***	1.01***	NS	−0.04*	−0.07***	NS	0.005***	0.004**	0.64***	0.01***	NS
R. melanophloeos	−7.12**	1.51***	0.01**	−0.05**	−0.07***	NS	NS	NS	NS	0.004***	***
E. arborea	−17.70***	NS	0.02***	NS	−0.02***	NS	0.004***	0.004**	0.53**	0.01***	***
Others	17.73***	NS	0.02***	NS	NS	NS	NS	NS	−0.33**	−0.01***	NS
PCT GI	30.17***	NS	0.04***	NS	NS	NS	NS	−0.003*	NS	−0.01***	*
PCT GII	−2.52	NS	0.05***	0.02*	NS	NS	NS	NS	0.77***	0.001	*
PCT Total	18.28***	NS	0.07***	NS	NS	NS	NS	NS	NS	−0.005***	**
MT GI	13.33***	1.16***	NS	−0.05*	−0.05*	NS	NS	−0.004**	NS	−0.002*	***
MT GII	−12.92***	0.98**	0.01**	−0.05*	−0.08***	NS	0.003**	0.004**	NS	0.01***	**
MT total	1.59	1.61***	0.02**	−0.07**	−0.08***	NS	NS	NS	NS	0.003***	*
PCT and MT total	10.38***	NS	0.08***	NS	NS	NS	NS	NS	NS	−0.001*	**

Notes: [a]Levels of significance: *$p < 0.1$, **$p < 0.05$, ***$p < 0.01$, NS indicates that the variable was not significant and that it was omitted from the final model.

stant. These differences were statistically significant ($p < 0.01$). For the broader tree categories, the results shown in the bottom part of Table 2 indicate that the densities of PCTs, both for GI and GII species and in total, did not differ significantly between the two management regimes. Conversely, considerable difference was observed for densities of MTs, where forest under participatory management had significantly higher total densities of MTs (GI and GII species and in total) than forest under government management.

For all species, except *P. viridiflorum*, the density varied significantly along altitudinal gradients; for some the density decreased with altitude and for others the density increased with altitude. The last column of Table 2 shows that the smoothing spline used to describe the nonlinear effect of altitude also played a statistically significant role for most species. The effects of altitude on density of PCTs and MTs of GI species was negative and significant ($p < 0.1$ *for MT GI and* $p < 0.01$ *for PCT in total and PCT GI*), while a positive effect of altitude was observed on MTs of GII species and for MTs in total *(*$p < 0.01$). This reflects the fact that most GI species are found at lower altitudes, while GII species are mostly found at higher altitudes. However, PCTs of GII species are distributed uniformly in all altitudinal ranges, with no significant effects of altitude. Because of the large effect of altitude on density of most species, we ran a separate model with altitude as the only explanatory variable, thus illustrating the variation of species density along altitudinal gradients. In Figure 3, this simple model is shown with the density observations and the values predicted by the models in Table 2.

Slope had a positive and significant effect on the density of some species, PCTs and MTs of both GI and GII species, but no negative effect was observed for any species or PCTs and MTs. East- and west-facing slopes have similar patterns of influence for most species and, when significant, they are mostly negatively related to density of individual species, PCTs and MTs.

For the structural variables, our results showed no relationship between forest size (FUG area) and density for the broader groups of trees, except for MTs of GII species where a positive significant effect of FUG area was observed. For individual species, FUGs with large area tended to have higher densities of some species, while for a single species (*A. falcatus*), the opposite effect was observed. Effects of the share of Ericaceous shrub land were mostly weak, but significant negative effects were observed for MTs and PCTs of GI species, and significant positive effects were detected both for the high altitude species *E. arborea* and *H. lanceolatum*. Similarly, a significant negative effect was observed for the low altitude species *J. excelsa*. The effect of distance to market was weak, but a significant positive relationship was observed for PCTs of GII species, showing that FUGs located farther from markets have higher densities

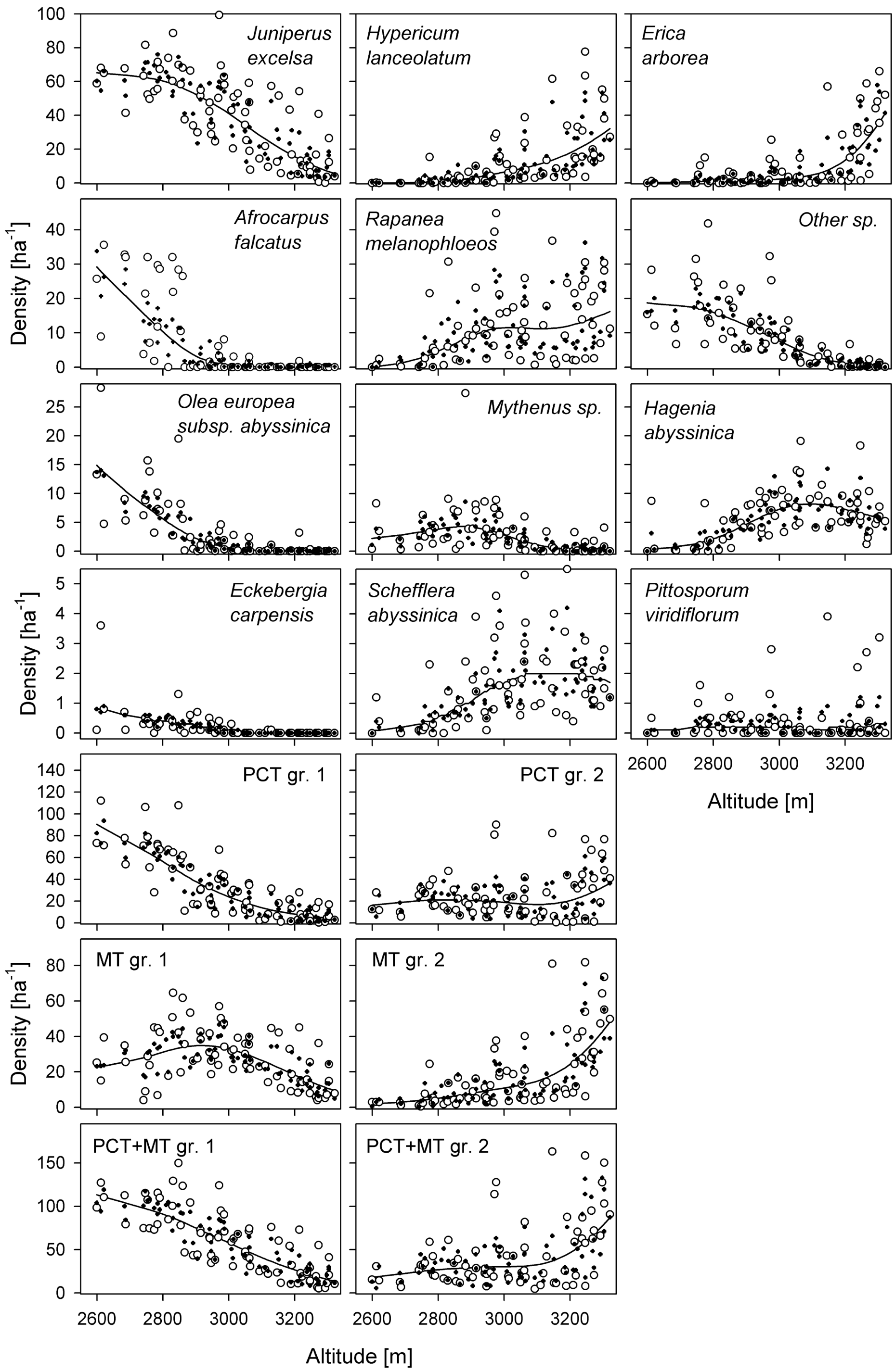

Figure 3. Generalized additive models and the altitudinal variation of densities of individual species and tree categories.
Notes: Symbols: open circles, observed values; dots, values predicted by final models (Table 2); unbroken curve, predictions by model with altitude only.

of these species. This agrees with the positive effects observed for the individual species *E. arborea* and *H. lanceolatum*, which are both in GII. In contrast, negative effects were observed for some of the species in GI: *H. abyssinica*, *O. europaea* and *Maytenus spp*.

3.2. PS matching

The probit model applied for estimating PS is attached in Appendix 1. The model predicts the probability of a forest being under participatory management and has a pseudo-R^2 of 52%. A summary of the balancing test for covariates used for matching the forests is presented in Appendix 2 that shows that the mean bias was reduced from 81 to 14, and the variables were not jointly significant (Pr > $\chi2$ = 0.122) after matching. All variables used in the matching models were balanced, and significant differences noted in the variables before matching between forests under the two management regimes disappeared

Table 3. Average treatment effect on treated (ATT) based on propensity score matching of forests under participatory management and forests under government management using Kernel and nearest-neighbour matching (NNM).

	Kernel (Epanichnikov) bw (0.06)		NNM (3 neighb.) Caliper (0.06)	
Density of	ATT (ha^{-1})	T-stat	ATT (ha^{-1})	T-stat
PCT GI	0.91 (16)	0.06	−3.13 (17)	−0.18
PCT GII	5.87 (11)	0.53	6.06 (11)	0.57
PCT total	6.80 (19)	0.37	2.94 (21)	0.14
MT GI	14.89 (7)**	2.24	15.72 (7)**	2.44
MT GII	10.35 (7)	1.44	11.35 (7)	1.53
MT total	25.24 (9)***	2.85	27.07 (10)***	2.83
PCT and MT	32.07 (15)**	2.09	30.04 (15)*	1.95
A. falcatus	7.7 (4.4)*	1.76	7.7 (4.6)*	1.67
J. excelsa	3.85 (12)	0.32	3.85 (11.7)	0.3
H. abyssinica	0.65 (1.7)	0.3	0.65 (2.1)	0.3
O. europaea	1.79 (2.1)	0.85	1.79 (2)	0.8
E. capensis	0.36 (0.4)	0.93	0.36 (0.4)	1.0
Maytenus spp.	−0.04 (1.8)	−0.02	−0.04 (1.8)	−0.02
S. Abyssinica	0.82 (0.4)**	1.98	0.82 (0.4)*	1.9
P. viridiflorum	−0.04 (0.4)	−0.09	−0.04 (0.5)	0.1
H. lanceolatum	11.35 (5.9)*	1.94	11.35 (6.1)*	1.86
R. melanophloeos	7.03 (3.6)*	1.93	7.03 (4.1)*	1.72
E. arborea	1.78 (9.5)	0.19	1.78 (10.7)	0.17
Others	−3.86 (6)	−0.64	−3.86 (5.9)	−0.66

Notes: Treatment group = forest under participatory management; Bootstrapped standard error (1000 replications) is shown in parenthesis. Levels of significance: *p < 0.10, **p < 0.05, ***p < 0.01.

after matching (detailed test results for each covariate is specified in Appendix 3).

Based on kernel and nearest neighbour matching, Average Treatment effects on the Treated group of forest are reported in Table 3. Both matching models dropped five FUGs from the group of forests under participatory management (Treatment) because of lack of common support as they were different from others with regard to the explanatory variables used for matching the forest under the two management regimes. Hence, in total, 69 FUGs were matched, 31 from the treated and 38 from the control group. The result from matching models (Table 3) shows that the differences in density of PCTs detected in Table 1, both for GI and GII species, were not statistically significant after controlling for confounding factors. The main difference found between groups was that the densities of MTs in GI and in total were significantly greater in forests under participatory management. Table 3 shows that the differences in density of MTs of GI species was 15–16 trees ha^{-1} ($p < 0.05$), higher for forests under participatory management than for government-managed forests. For MTs in total, the difference was 25–27 trees ha^{-1} ($p < 0.01$); and for PCTs and MTs, it was 32–30 trees ha^{-1} ($p < 0.1$). The species-wise comparison showed that the densities of four species (*A. falcatus, H.lanceolatum, R. melanophloeos* and *S. abyssinica*) appeared to be higher (p < 0.1 or better) in forest under participatory management compared with forest under government management. Most of the results of the matching exercise are similar to the results obtained using the GAM (Table 2), but differ considerably from the unadjusted estimates summarized in Table 1.

We compared the characteristics of the five FUGs that were dropped by the matching algorithm with FUGs in the matched treatment group (participatory management) based on the explanatory variables used for matching. The comparison showed that the omitted FUGs are generally located farther from the market, do not include areas at lower altitudes (≤2800 m) and are located on steep slopes. The densities of trees in the omitted FUGs, however, did not differ from densities in the matched FUGs.

3.3. Species composition

The distribution of species in the forest is heterogeneous, with most species observed only in few plots and with low average densities (Figure 4). Of the 11 individual species and the group of species classified as 'Others', the four most abundant species in the forest were *J. excelsa* (relative density 38%), *R. melanophloeos* (12%), *H. lanceolatum* (12%) and *E. arborea* (10%). These four species constituted 72% of the total stem number (abundance), but due to their altitudinal distribution (Figure 4), and possibly clustering, their total relative frequency was lower (53%). Together, the three major timber species *J. excelsa, A. falcatus and H. abyssinica* made up 49% of the total density (relative density) and 39% of the total relative frequency. *R. melanophloeos, H. lanceolatum and E. arborea*, the three dominant upper altitude species, constituted 39% of the relative density and 35% of the relative frequency.

The relative frequency of *J. excelsa* was lower (21%) than the relative density (38%), reflecting that *J. excelsa* was abundant but only occurred at lower altitudes (Figure 4). By contrast, *H. abyssinica* occurred only at higher altitudes and was much less abundant than *J. excelsa*. This implies that the relative density of *H. abyssinica* was low, but its relative frequency was only slightly lower than that of *J. excelsa* and comparable to that of *R. melanophloeos*, which had higher density. *A. falcatus*, which was

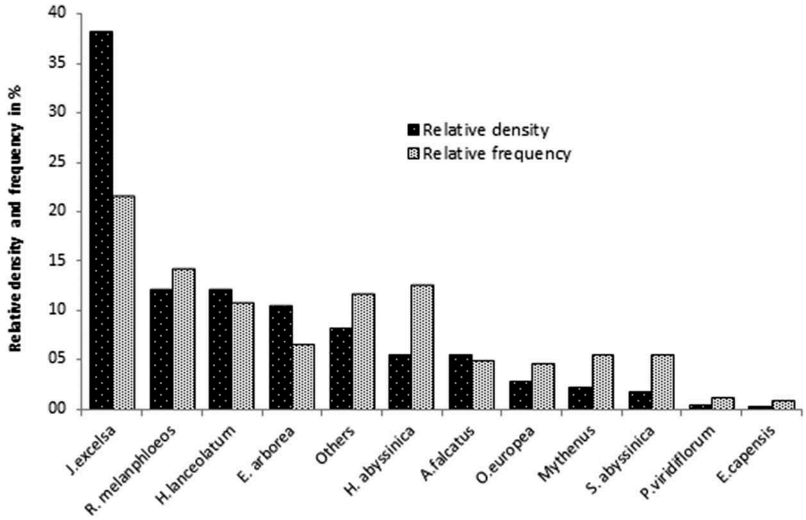

Figure 4. Relative density and relative frequency of species.

found only at lower altitudes, had the same relative density and frequency (5%).

4. Discussion

4.1. Influence of topographical and structural features

Our findings that altitude and slope were the topographical variables that had the strongest influence on species distribution in the forest agree with several other studies (e.g. Leathwick & Austin 2001; Agrawal & Chhatre 2006; Bader & Ruijten 2008; Somanathan et al. 2009). In part, the observed positive effect of steep slopes on the density of many tree species and size categories are likely due to the difficulty of accessing and harvesting trees on steep slopes and the limited suitability of sloping terrain for cultivation. The fact that the effect of slope was not significant for MTs of GI species may indicate that MTs in this group have large diameters, mostly above 80 cm and with large crowns that make them prone to damage by wind and landslides in areas with steep slopes. Asrat et al. (1997) also reported that soils on steep slopes and at the top of ridges in the area are shallow. The negative influence of east- and west-facing slopes on the densities of some species may be due to the fact that these slopes are drier than the north/south facing slopes because of the amounts of solar energy intercepted. In the tropics, north-south differences are less pronounced, whereas east-west differences may be larger than in temperate regions (Sarmiento 1986).

The size of the user group relative to FUG forest area has been suggested as an important explanatory variable for changes in forest conditions with the establishment of PFM (Nagendra 2007). However, forest area and number of households per hectare did not appear to influence the density of trees in the present study. This is not surprising as the number of members in a FUG in Dodola is limited to 30 households and as the area of forest per FUG is adjusted to render sustainable forest management possible given the current distribution of settlements in the forest. Previous studies have shown that inter-group and intra-group inequality can affect the ecological outcomes of PFM (Ostrom & Nagendra 2006). Since our purpose was not to make comparisons among FUGs, we have not considered such variables. However, a previous study in the same area (Rustagi et al. 2010) shows that FUGs with high shares of conditional co-operators (an individual cooperation being conditional on the cooperation of others) are characterized by higher densities of PCTs than FUGs with low shares of conditional co-operators. The same study also documented that other socio-demographic variables, such as education and clan difference, did not affect conservation outcomes of PFM significantly.

4.2. Impact estimates

When the inventory data used in the present study was collected (2004–2005), PFM had been in operation for only 3–5 years. The estimated bias adjusted differences (Table 3) in densities of trees between forest under PFM, and government management indicate that, given the topographical and structural variables available for correcting the estimates and under the conditions prevailing at the site, PFM was apparently capable of protecting the forest. The original, biased density differences (Table 1) showed that out of 11 tree species, the densities of nine individual species, PCTs and MTs were higher in forest under participatory management (p < 0.01) than in forest under government management. After controlling for confounding variables, however, only the densities of four species and MTs were higher and significantly different from forest under government management. Our findings agree with other studies that have applied similar approaches to analyse the effects of protected areas. For instance, Andam et al. (2010) showed that estimates of deforestation reduction rates in protected areas decreased from 65% to 10% when controlling for variables causing bias. Given the short time that PFM had been in operation when the data were collected, it appears surprising that differences between PFM and government forest could be detected. Partly this may be a consequence of FUGs actually protecting PFM forest, but it may also be related to leakage, increasing extraction from government forest. Moreover, it is also possible that the set of topographical, geographical and structural variables that we controlled for was insufficient. For example, we cannot rule out that initial forest quality (stocking) had at least some impact on the likelihood that a patch of forest was selected for PFM.

Although we did not observe differences in densities of PCTs between the forests under the two management regimes, visual observation in the forest and discussion with key informants during field work in 2010 indicated that regeneration is more abundant in the forest where PFM was established in the first phase compared to areas where PFM was established later. A possible explanation for the lack of difference in PCTs may be the short time period of the study; three to five years does not allow newly established trees to grow into the categories of trees that were measured (PCT and MT). Besides, the criterion followed during the inventory, that when dense patches of young trees were encountered the minimum distance between PCTs recorded should be 4 m, potentially leads to underestimation of the density of PCTs, particularly in FUGs established in the first phase if these did manage to protect the regeneration.

The GAM and the two matching models yielded consistent results, showing the robustness of our estimates. When developing the models, we tried out all topographical and structural variables that were available, carried out structured variable selection and checked that after matching no difference remained between forest categories on the chosen variables (Table 4 and Appendices 2 and 3). However, although we captured most effects of confounding variables, it cannot be ruled out that a limited bias may remain due to limitations on the set of covariates available for matching the two groups. For example, variables such as forest stocking in 1999–2000 or population density within three hours walking distance could have been included. We would, however, not expect the inclusion of such variables to change the outcome considerably.

Table 4. Balancing test of covariates used for matching (kernel and NNM methods) forests under participatory management and forests under government management.

Sample	Pseudo R^2	LR χ^2	$p>\chi^2$	Mean Bias	Median Bias
Unmatched	0.519	53.23	0.000	81.4	55.8
Matched (Kernel)	0.117	10.06	0.122	14.1	14.1
Matched (NNM)	0.117	9.5	0.147	16.5	19.1

The forest stand structure shows that all parts of the forest, particularly those at lower altitude, are dominated by a single species, *J. excelsa*. The analysis showed that the density of trees in the middle diameter class was low for all species, presumably due to past selective cutting, thus calling for special attention to the management of trees in this diameter class (Amente et al. 2006).

4.3. Is there a leakage effect of PFM?

Our finding that higher densities of MTs (particularly GI) were observed in forest under participatory management compared to forest under government management may actually indicate that leakage had occurred. A study by Tadesse (1999) has shown that the annual deforestation rate in the area between 1993 and 1997, before establishment of PFM, was 3%. Between 2002 and 2006, after PFM was introduced in the area, the forest cover under PFM increased by 15.6% (4% yearly), while the forest cover of adjacent areas under government management decreased by almost the same amount, 16% (4% yearly) (Neuman 2008). Using the 3% annual deforestation rate reported by Tadesse (1999) as a reference, the added net annual deforestation rate in forest under government management that are located adjacent to PFM forests is then estimated at 1%. If other factors (market, logistics and population) remained unchanged between 1993–1997 and 2002–2006, we may assume that the seeming net increase in the degradation rate in government-managed forest is mainly a result of leakage effects of PFM. Considering the annual increase in forest cover under PFM was about 4% between 2002 and 2006 and the areas of FUGs under PFM and government management were 13,375 ha and 9671 ha, respectively, the net gain caused by PFM is more than the loss. Where PFM was introduced, the extraction of forest products has partly been displaced to forest areas under government management. It means that the institutional arrangements of participatory management (the introduced limits on the size of user groups, restrictions on the number of MTs that a user group may harvest and on the expansion of agricultural and homestead areas in PFM forests, promotion of ecotourism and expansion of tree planting activities inside and outside the forest) have worked. In addition, the positive net effect of PFM on forest conservation seems higher than the loss related to leakage, indicating that PFM is a useful forest conservation approach.

5. Conclusion

This article contributes to the literature on impacts of decentralized forest management on forest conditions by comparing observed densities of different tree species and size categories in forests under participatory and government management. As opposed to most previous impact studies that were based on macro-level data (Coleman & Fleischman 2012), the main contribution of our analysis is to provide a micro-level analysis.

Results from this analysis show that the overall densities of MTs ha^{-1} and four individual species (*Afrocarpus falcatus Schefflera abyssinica, Hypericum lanceolatum* and *Rapanea melanophloeos*) were higher in forests under participatory management ($p < 0.01$). The noted differences in vegetation densities between the two management regimes indicate that, when controlling for topographical and structural confounding variables, PFM does actually appear to contribute positively to the distribution and density of different species in the forest. The effects of confounding variables may be large, and our study suggests that the effects of topographical variables need to be considered when comparing the impacts of different management regimes. The present study supports previous arguments that compared to other types of forest management regimes in developing countries, decentralized management has a potential to reduce deforestation rates (e.g. Somanathan et al. 2009; Coleman & Fleischman 2012; Porter-Bolland et al. 2012).

Acknowledgements

We are grateful for Dodola District Forest Enterprise Office for permission to use the forest inventory data. The first author in particular is very thankful to Zerihun Dheko, who provided the required information and assisted the field work. We thank two anonymous referees for their valuable comments.

Disclosure statement

No potential conflict of interest was reported by the authors.

Funding

This work was supported by the University of Copenhagen.

ORCID

Aklilu Ameha http://orcid.org/0000-0002-7698-251X

References

Agrawal A, Chhatre A. 2006. Explaining success on the commons: community forest governance in the Indian Himalaya. World Dev. 34:149–166.

Ameha A, Larsen HO, Lemenih M. 2014a. Participatory forest management in Ethiopia: learning from pilot projects. Environ Manage. 53:838–854.

Ameha A, Nielsen OJ, Larsen HO. 2014b. Impacts of access and benefit sharing on livelihoods and forest: case of participatory forest management in Ethiopia. Ecol Econ. 97:162–171.

Amente G, Huss J, Tennigkeit T. 2006. Forest regeneration without planting: the case of community managed forests in the Bale mountains of Ethiopia. J Drylands. 1:1.

Andam KS, Ferraro PJ, Sims KRE, Healy A, Holland MB. 2010. Protected areas reduced poverty in Costa Rica and Thailand. Proc Natl Acad Sci. 107:9996–10001.

Angelsen A, Wunder S. 2003. Exploring the forest–poverty link: key concepts, issues and research implications. Cifor Occasional Paper Number 40. Bogor (Indonesia): Center for International Forestry Research. Avilabale from: Http://www.Cifor.Cgiar.Org/publications/papers.

Asrat A, Kebede S, Geremew Z. 1997. Study report on the geology and pedology of Dodola area. Dodola (Ethiopia): IFMP. Unpublished.

Bader MY, Ruijten JJA. 2008. A topography-based model of forest cover at the alpine tree line in the tropical andes. J Biogeogr. 35:711–723.

Becker SO, Ichino A. 2002. Estimation of average treatment effects based on propensity scores. Stata J. 2:358–377.

Bowler D, Pullin AS. 2010. The Evidence Base for Community Forest Management as a Mechanism for Supplying Global Environmental Benefits and Improving Local Welfare: A Stap Advisory Document. [cited 2012 Jun 28]. Available from: Http://www.Stapgef.Org/evidence-base-for-cfm.

Brown DG. 1994. Predicting vegetation types at treeline using topography and biophysical disturbance variables. J Vegetation Sci. 5:641–656.

Caliendo M, Kopeinig S. 2008. Some practical guidance for the implementation of propensity score matching. J Econ Surv. 22:31–72.

Caplow S, Jagger P, Lawlor K, Sills E. 2011. Evaluating land use and livelihood impacts of early forest carbon projects: lessons for learning about redd+. Environ Sci Policy. 14:152–167.

Chakraborty RN. 2001. Stability and outcomes of common property institutions in forestry: evidence from the terai region of Nepal. Ecol Econ. 36:341–353.

Coleman EA, Fleischman FD. 2012. Comparing forest decentralization and local institutional change in Bolivia, Kenya, Mexico, and Uganda. World Dev. 40:836–849.

Dessie G, Christiansson C. 2008. Forest decline and its causes in the south-central rift valley of Ethiopia: human impact over a one hundred year perspective. AMBIO: J Hum Environ. 37:263–271.

Earth Trends. 2007. Forests, grasslands, and drylands: country profiles-Ethiopia. Earth trends/world resources institute. [cited 2012 Apr 7]. Available from: Http://earthtrends.Wri.Org/country_profiles

Ferraro PJ. 2009. Counterfactual thinking and impact evaluation in environmental policy. New Dir Eval. 2009:75–84.

Ferraro PJ, Hanauer MM, Sims KRE. 2011. Conditions associated with protected area success in conservation and poverty reduction. Proc Natl Acad Sci. 108:13913–13918.

Ferraro PJ, Pattanayak SK. 2006. Money for nothing? A call for empirical evaluation of biodiversity conservation investments. PLoS Biol. 4:482–488.

Fichtmüller S. 1997. Useful plants in the vicinity of Dodola [Unpublished]. Dodola (Ethiopia): IFMP.

Friis I. 1992. Forests and forest trees of northeast tropical africa: their natural habitats and distribution patterns in

Ethiopia, Djibouti and Somalia. Kew bulletin additional series, volume 15. London (UK): Royal Botanic Gardens, Kew.
Ganuza A, Almendros G. 2003. Organic carbon storage in soils of the basque country (spain): the effect of climate, vegetation type and edaphic variables. Biol Fertil Soils. 37:154–162.
Gautam AP, Shivakoti GP, Webb EL. 2004. Forest cover change, physiography, local economy, and institutions in a mountain watershed in Nepal. Environ Manage. 33:48–61.
Gobeze T, Bekele M, Lemenih M, Kassa H. 2009. Participatory forest management and its impacts on livelihoods and forest status: the case of Bonga forest in Ethiopia. Int For Rev. 11:346–358.
Guisan A, Edwards TC Jr, Hastie T. 2002. Generalized linear and generalized additive models in studies of species distributions: setting the scene. Ecol Modell. 157:89–100.
Husch B, Beers TW, Kershaw JA. 2003. Forest mensuration. Hoboken (NJ): John Wiley & Sons; p. 456.
Johansson MU, Fetene M, Malmer A, Granström A. 2012. Tending for cattle: traditional fire management in Ethiopian Montane Heathlands. Ecol Soc. 17:article 19
Kubsa A, Mariame A, Amente G, Lipp HJ, Tadesse T. 2003. WAJIB: an alternative forest conservation approach for Ethiopia's forest. Xii World Forestry Congress; Quebec (Canada). [cited 2011 Nov 5]. Available from: Http://www.Fao.Org/docrep/article/wfc/xii/0145-c2.Htm.
Lamprecht H. 1989. Silviculture in the tropics. Tropical forest ecosystems and their tree species-possibilities and methods for their long-term utilization. Eschborn (Gemany): Gtz.
Lawrence A. 2007. Beyond the second generation: towards adaptiveness in participatory forest management. CAB Rev Perspect Agric Vet Sci Nutr Nat Resour. 2:1–15.
Leathwick JR, Austin MP. 2001. Competitive interactions between tree species in newzealand's old-growth indigenous forests. Ecology. 82:2560–2573.
Leuven E, Sianesi B. 2003. Psmatch2: Stata module to perform full mahalanobis and propensity score matching, common support graphing, and covariate imbalance testing. Statistical Software Components. S432001. Chestnut Hill (MA): Boston College.
Limenih M, Temesgen Z. 2011. History and experiences of PFM in Ethiopia: capturing lessons learnt and identifying gap [Unpublished]. Addis Ababa: Federal Democratic Republic of Ethiopia, Ministry of Agriculture and Natural Resources - participatory forest management up scaling.
Lund JF, Balooni K, Casse T. 2009. Change we can believe in? Reviewing studies on the conservation impact of popular participation in forest management. Conserv Soc. 7:71–82.
Nagendra H. 2007. Drivers of reforestation in human-dominated forests. Proc Natl Acad Sci. 104:15218–15223.
Neuman M 2008. Policies to increase forest cover in Ethiopia. In: Bane J, Nune S, Mekonnen A, Bluffstone R editors. Proceedings of a policy workshop organized by Environmental Economics Policy Forum for Ethiopia (EEPFE) and Ethiopian Development Research Institute (EDRI); 2007 Sep 18–19; Addis Ababa (Ethiopia). [cited 2013 Jan 25]. Available from: http://www.efdinitiative.org/research/publications/publications-repository/forest-cover-in-Ethiopia-workshop-proceedings
Ostrom E, Nagendra H. 2006. Insights on linking forests, trees, and people from the air, on the ground, and in the laboratory. Proc Natl Acad Sci. 103:19224–19231.
Oyono PR, Biyong MB, Samba SK. 2012. Beyond the decade of policy and community euphoria: the state of livelihoods under new local rights to forest in rural Cameroon. Conserv Soc. 10:173–181.
Pfaff A, Robalino J, Lima E, Sandoval C, Herrera LD. 2014. Governance, location and avoided deforestation from protected areas: greater restrictions can have lower impact, due to differences in location. World Dev. 55:7–20.
Poffenberger M. 2006. People in the forest: community forestry experiences from Southeast Asia. Int J Environ Sustainable Dev. 5:57–69.
Porter-Bolland L, Ellis EA, Guariguata MR, Ruiz-Mallén I, Negrete-Yankelevich S, Reyes-García V. 2012. Community managed forests and forest protected areas: an assessment of their conservation effectiveness across the tropics. For Ecol Manage. 268:6–17.
Ribot JC, Agrawal A, Larson AM. 2006. Recentralizing while decentralizing: how national governments reappropriate forest resources. World Dev. 34:1864–1886.
Rosenbaum PR. 2002. Observational studies. Newyork: Springer.
Rosenbaum PR, Rubin DB. 1983. The central role of the propensity score in observational studies for causal effects. Biometrika. 70:41–55.
Rustagi D, Engel S, Kosfeld M. 2010. Conditional cooperation and costly monitoring explain success in forest commons management. Science. 330:961–965.
Sarmiento G. 1986. Ecologically crucial features of climate in high tropical mountains. In: Vuilleumier F, Monasterio M, editor. High altitude tropical biogeography. Oxford: Oxford University Press; p. 11–45.
Schmitt J. 2003. The contribution of community forest management to the rural household economy: A case study from bale mountains, Ethiopia. I-Too Working Paper No. 11. Freiburg: University of Freiburg; p. 73.
Schreckenberg K, Luttrell C. 2009. Participatory forest management: A route to poverty reduction? Int For Rev. 11:221–238.
Sianesi B. 2004. An evaluation of the swedish system of active labor market programs in the 1990s. Rev Econ Stat. 86:133–155.
Somanathan E, Prabhakar R, Mehta BS. 2009. Decentralization for cost-effective conservation. Proc Natl Acad Sci. 106:4143–4147.
Stipl P. 1988. Report on plant sociology survey in Adaba-Dodola NFPA [Unpublished]. Dodola (Ethiopia): IFMP.
Sunderlin WD, Angelsen A, Belcher B, Burgers P, Nasi R, Santoso L, Wunder S. 2005. Livelihoods, forests, and conservation in developing countries: an overview. World Dev. 33:1383–1402.
Tadesse T. 1999. A new look towards the conservation of the Adaba-Dodola forest [Unpublished]. Dodola (Ethiopia): IFMP; p. 13.
Takahashi R, Todo Y. 2012. Impact of community-based forest management on forest protection: evidence from an aid-funded project in Ethiopia. Environ Manage. 50:396–404.
Tesfaye Y, Roos A, Campbell BM, Bohlin F. 2011. Livelihood strategies and the role of forest income in participatory-managed forests of Dodola area in the bale highlands, Southern Ethiopia. For Policy Econ. 13:258–265.
Uhlig SK, Uhlig K. 1991. Studies on the altitudinal zonation of forests and alpine plants in the central bale mountains, Ethiopia. Mt Res Dev. 11:153–156.
USGS. 2013. Earth Explorer Gdem Data. [cited 2013 Feb 10]. Available from: Http://glovis.Usgs.Gov.
Yimer F, Ledin S, Abdelkadir A. 2006. Soil property variations in relation to topographic aspect and vegetation community in the south-eastern highlands of Ethiopia. For Ecol Manage. 232:90–99.

Appendix 1

Parameter estimates of probit model used to estimate propensity scores.

Variables	Estimates	t-stat
Walking distance to market (hours)	0.942**	−2.39
Area with altitude ≤2800 m (%)	0.0176*	−1.88
Area with altitude >3100 m (%)	−0.02	−1.59
Area with slope ≤10° (%)	−0.002	−0.04
Area with slope >15° (%)	0.045	−1.64
Area east-west oriented (%)	−0.097**	−2.12
Intercept	3.62	−0.77
Log pseudo likelihood	−24.65	
Pseudo R^2	0.52	
Observations	74	

Response: 1 = forest under participatory management, 0 = forest under government management.
Levels of significance: * $p < 0.10$, ** $p < 0.05$, *** $p < 0.01$.

Appendix 2

Balancing test for Kernel matching (bw 0.06).

Variable	Matched	PFM Forest	Gov. forest	%bias	% redun. |bias|	t	$p > t$
Market distance in walking hr.	Before	2.37	2.00	46		1.97	0.05
	After	2.27	2.15	14.9	67.6	0.55	0.59
Percentage of area with altitude ≤2800 m	Before	27.10	8.51	65.6		2.84	0.01
	After	31.38	25.68	20.1	69.3	0.62	0.54
Percentage of area with altitude >3100	Before	31.33	48.00	−43		−1.84	0.07
	After	26.40	21.24	13.3	69.1	0.56	0.58
Percentage of area with Slope ≤10 degrees	Before	14.15	32.48	−153.7		−6.54	0.00
	After	14.13	11.78	19.7	87.2	1.70	0.10
Percentage of area with Slope >15 degree	Before	65.64	34.20	165		7.05	0.00
	After	65.38	67.45	−10.9	93.4	−0.61	0.54
Percentage of area East/west oriented	Before	79.88	80.75	−15.2		−0.66	0.51
	After	80.24	80.56	−5.8	62.3	−0.19	0.85

Appendix 3

Balancing test for nearest neighbour matching with 3 neighbours and caliper 0.06.

Variable	Matched	PFM Forest	Gov. forest	%bias	% redun |bias|	t	$p > t$
Market distance in walking hr.	Before	2.37	2.00	46.00		1.97	0.05
	After	2.27	2.10	21.00	54.3	0.80	0.43
Percentage of area with altitude ≤2800 m	Before	27.10	8.51	65.60		2.84	0.01
	After	31.38	30.40	3.50	94.7	0.10	0.92
Percentage of area with altitude >3100	Before	31.33	48.00	−43.00		−1.84	0.07
	After	26.40	18.95	19.20	55.3	0.82	0.41
Percentage of area with Slope ≤10 degrees	Before	65.64	34.20	165.00		7.05	0.00
	After	65.38	67.86	−13.00	92.1	−0.73	0.47
Percentage of area with Slope >15 degree	Before	14.15	32.48	−153.70		−6.54	0.00
	After	14.13	11.87	18.90	87.7	1.64	0.11
Percentage of area East/west oriented	Before	79.88	80.75	−15.20		−0.66	0.51
	After	80.24	81.55	−23.20	−52.2	−0.76	0.45