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Original Research Article

Impact of biogas interventions on forest biomass and regeneration in southern India

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ABSTRACT

Programs to provide alternative energy sources such as biogas improve indoor air quality and potentially reduce pressure on forests from fuelwood collection. This study tests whether biogas intervention is associated with higher forest biomass and forest regeneration in degraded forests in Chikkaballapur district in Southern India. Using propensity score matching, we find that forest plots in proximity to villages with biogas interventions (treatment) had greater forest biomass than comparable plots around villages without biogas (control). We also found significantly higher sapling abundance and diversity in treatment than control plots despite no significant difference in seedling abundances and diversity in treatment forests, suggesting that plants have a higher probability of reaching sapling stage. These results indicate the potential for alternative energy sources that reduce dependence on fuelwood to promote regeneration of degraded forests. However, forest regrowth is not uniform across treatments and is limited by soil nutrients and biased towards species that are light demanding, fire-resistant and can thrive in poor soil conditions.

1. Introduction

Biomass is the fourth largest source of energy in the world, and in many countries, 90% of the total energy comes from traditional fuels such as wood, straw and dung (Hall, 1997). While fuelwood as an energy source has the advantage of being renewable and accessible to even the most marginalized, it also has disadvantages such as contributing to forest degradation, carbon and methane production from burning, and health hazards from household air pollution (Bluffstone et al., 2013; World Health Organization, 2014). Many programs aim to reduce consumption of fuelwood through providing alternative energy sources and efficient cooking stoves (International Energy Agency, 2016). These programs have met with mixed

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success, with some programs leading to reduced carbon emissions from burning (Feng et al., 2009; Garfí et al., 2012; Katuwal and Bohara, 2009).

Reduction in fuelwood use also theoretically allows forests to recover. Possible benefits for forests of programs that provide alternatives to fuelwood include increases in biomass, carbon sequestration, and recovery of species composition that enable the sustainable provision of ecosystem services in the future. Studies have examined the impact of alternatives to fuelwood and efficient cooking devices on tree selection by local people (Timko and Kozak, 2016), and the role of plantations and agroforestry on degraded lands to meet fuelwood demands and sequester carbon (Gruenewald et al., 2007; Khamzina et al., 2012). Yet, we could not find any study that explicitly examined forest recovery following an intervention to reduce fuelwood use. Studies on forest recovery examine recovery from mining, agriculture (Ruiz-Jaen and Aide, 2005) and other land uses and extraction activities (Jones and Schmitz, 2009), but do not examine recovery from fuelwood extraction. While recovery can be assisted (active restoration) or unassisted (passive restoration) (Chazdon, 2008), it is also important to examine 'incidental' restoration, where recovery is an unintended consequence of an intervention as these may follow trajectories different from the already complex trajectories of recovery (Aronson and Galatowitsch, 2008; Jacquet and Prodon, 2009; Matthews et al., 2009; Suding, 2011). Studies that examine impact of biogas and other initiatives to reduce fuelwood wood demand on forest growth and regeneration, preferably using longitudinal field studies that include surveys taken prior to and after an intervention (Baylis et al., 2016), will help establish patterns and understand trajectories. However, such studies take time, and policy-relevant answers are urgently needed.

This study used an existing biogas intervention, where the intervention was first implemented in 2005 (ten years prior to the study), to assess the impact of the intervention on forest recovery. In this study, we tested whether implementation of a biogas intervention in communities that previously harvested wood from nearby forests was associated with (1) higher biomass; (2) higher regeneration, and (3) different species composition than comparable forests where such an intervention did not take place. We also investigated other factors associated with forest recovery, including biophysical, socio-economic and landscape variables. Such a study is of particular importance in India, where restoration of degraded forests constitute an important component of its Nationally Determined Contribution (NDC) to the United Nations Framework Convention on Climate Change (UNFCCC) (Government of India, 2015), but the study results suggest a universal response that might be of global significance.

2. Materials and methods

2.1. Study site

The study was located in Chikkaballapur district in the state of Karnataka, India (Fig. 1 and 13.4324° N, 77.7280° E). This region is arid and receives an average of 731 mm of rainfall annually, 69% of which is from June to October. Population density is high in the district (298 individuals per km^2), and 88% of people are landless and dependent on work as agricultural laborers (Karnataka Forest Department, 2014). Around 40% of landholdings are <0.5 ha, while another 46% of landholdings are between 0.5 and 2 ha (Government of India, 2011).

Forests in the study area consist of dry deciduous and scrub forests on soils that are derived primarily from a bedrock of hornblende schist, although outcrops of granitic gneiss, laterite and dolomite are irregularly distributed in the landscape. The Forest Department Working Plan reports that heavy use for fuelwood and charcoal production in the past, as well as low rainfall and poor and shallow soil, has led to low biomass forests and branchy, stunted trees with diffused crowns. Common overstory species in the area include *Albizia amara*, *Cassia fistula*, and *Anogeissus latifolia*, and the understory consisted of *Lantana camara*. Poor regeneration of indigenous species such as *Hardwickia binata*, *Chloroxylon swietenia*, *Semecarpus anacardium* and *Cassia fistula* in these forests led the Forest Department to develop plantations of *Eucalyptus* species, *Prosopis juliflora*, *Senna siamea*, *Dalbergia sissoo* and *Casuarina equisetifolia* in the district (Karnataka Forest Department, 2014).

The forests in the district constitute 17% of the total land area, and are officially managed by the Forest Department. Forests are mostly located on hilly areas and have steep slopes that are inaccessible for management. The two largest blocks, or management units, in the study area are Narasimhadevarabetta (~160 km^2) (henceforth, NDB) and Ittikaldurga (~100 km^2) (henceforth, IKD), and it is unclear how long the forests in these blocks had been isolated from each other. The Forest Department also restricts many activities in the forests: grazing and collection of grass to stall-feed cattle is allowed in some forest areas; and the Forest Department Working Plan reports that grazers set fire to the forest to augment production of grass. These forests are also home to wildlife such as leopards, blackbuck, chital, cobras, porcupine, black-naped hares and rodents (Karnataka Forest Department, 2014).

Biogas digesters were introduced in the area in 2005 with the aim of replacing fuelwood in 885 villages. The project itself constituted an agreement between French company VELCAN Energy and a village-level social organization called the Coolie Sangha in 2005 – a 39 year old organization of smallholder farmers from the villages in the district – and between Dutch company Fair Climate Fund and the Coolie Sangha in 2008. The local NGO facilitating the project installed 16,682 biogas units in villages where members of the Coolie Sangha were enthusiastic about using biogas interventions to solve their fuelwood problems. Individual families took a loan from the village-level Coolie Sangha to purchase the biogas units.

These biogas units were underground composters that used cattle dung to generate gas that was piped to a kitchen stove and used for cooking. A comparison of households with and without biogas units in the area identified that households with biogas units had improved diets and time allocation (Anderman et al., 2015). Due to absence of historical reference sites or undisturbed references sites, our strategy was to compare sites close to interventions against sites without interventions, which could be considered unrestored degraded sites (Suding, 2011). Because biogas units were distributed in clusters, a pattern of areas with and without biogas interventions created a natural experiment wherein we could match sites to test for impact on forest composition, aboveground biomass and regeneration. Ecosystem recovery can take from ten years (Aronson and Galatowitsch, 2008; Jones and Schmitz, 2009) to 30–40 years (Chazdon, 2008). In the test area, ten years had passed since the biogas intervention and it was reasonable to expect that this was sufficient for some changes to be evident in the forest. Lack of detectable improvement would also have been an interesting result.

2.2. Data and analysis

We collected information on various factors that could influence forest recovery (section 2.3). We then divided the study area (~260 km^2) into 5 × 5 km grids and identified comparable treatment and control grid units, where each unit contained a cluster of villages with or without existing biogas interventions (section 2.4), and then measured ecological parameters in two plots in each grid unit (section 2.5). Although adjacent control and treatment units were matched at the time of sampling (pair-wise sampling, ten years after biogas project was implemented), we also used propensity score matching (henceforth, PSM) to independently match grid units (section 2.6) and used both the original pair-wise sampling and PSM to analyze the data (section 2.7).

2.3. Data collected

We collected data on biophysical, demographic, socio-economic and management variables that may influence impact on forests (sources in Table 1) so that we could account for variation due to these factors in our study design.

For biophysical variables such as temperature and precipitation, we resampled data from MODIS 11A1 and TRMM 3B43 to 30-meter resolution to make it comparable with our other data. For fire frequency, we expected to use MODIS 14A1, and calculate mean annual fire radiative power (FRP) for each year, but MODIS FRP did not capture any incidences of fire from 2000 to 2016 in our study area, and we had to rely on our field observations for fire (detailed in section 2.5).

For demographic variables, we calculated pressure from livestock (cattle and buffaloes) and small ruminants (goat and sheep) separately, as grazing and browsing have different impacts on the forest. We calculated human population pressure on the forest from human populations. For all three, we used inverse kriging to interpolate village-level populations of livestock, small ruminants and people to a maximum distance of 5 km (Karnataka Forest Department, 2014 reports that the zone of influence of a village extends to 5 km). Inverse kriging to 5 km allows overlap of zones of influence of different villages, which represents on-ground forest use patterns better than assuming that the forest is used only by the adjacent village. To estimate market pressure, we first identified towns in the landscape (classified by the government as managed by municipal

Table 1
Potential factors associated with forest recovery.

Factor	Source	Spatial Resolution
Biophysical Variables		
Mean Annual Temperature	MODIS 11A1 (reverb.echo.nasa.gov)	1 km × 1 km
Mean Precipitation in the Wettest Quarter	TRMM 3B43 (trmm.gsfc.nasa.gov)	0.25° × 0.25°
Mean Annual Fire Radiative Power (2000–2016)	MODIS 14A1 (reverb.echo.nasa.gov)	1 km × 1 km
Fire Signs	Field surveys	Plot
Elevation	ASTER DEM (reverb.echo.nasa.gov)	30 m × 30 m
Slope	ASTER DEM (reverb.echo.nasa.gov)	30 m × 30 m
Soil characteristics	Field surveys	Plot
Demographic Variables		
Population (Population density per pixel)	(Census of India, 2011)	Village
Livestock (Density of cows and buffaloes per pixel)	Livestock Census (https://data.gov.in/catalog/details-livestock-18th-livestock-census)	Village
Small Ruminants (Density of goat and sheep per pixel)	Livestock Census (https://data.gov.in/catalog/details-livestock-18th-livestock-census)	Village
Socio-economic variables		
Market pressure (Town population impact per pixel)	(Census of India, 2011)	Village
Distance to roads (Distance to roads per pixel)	Forest Department road shapefile	Village
Forest Management variables		
Forest Block (NDB or IKD)	Forest Department	30 m × 30 m
Forest type	Forest Department	30 m × 30 m
Energy variables		
Proportion of households with biogas units in each village	Local NGO	Village
Proportion of households with LPG gas connections	LPG census, 2011 (censusindia.gov.in/2011census/hlo/HLO_Tables.html)	Village
Vegetation variables		
Canopy Cover	Field surveys	Plot
Species Composition and Shannon-Wiener Diversity Index	Field surveys	Plot

corporations). Because market pressure is not limited by distance, we calculated market pressure by inverse kriging the population of towns in the area without using any limits for maximum distance. We used ArcGIS (9.3, Environmental Systems Research Institute, Redlands, California, USA) for all interpolations.

We used shapefiles provided by the Karnataka Forest Department to categorize forest type and forest blocks (as authorities responsible for management differed between the blocks) (Fig. 1), and excluded forest plantations from our potential sampling areas. For villages where the biogas intervention took place, not all households had biogas. Therefore, we used data from the NGO that was responsible for biogas intervention to quantify proportion of households where biogas was installed. Because use of liquefied petroleum gas (henceforth, LPG) gas could confound our analysis as households using LPG would be less dependent on the forest, we also quantified proportion of households with LPG connection in each village. We used inverse kriging to interpolate proportion of households with biogas and proportion of households with LPG up to a distance of 5 kilometers from each village.

2.4. Sampling design

From the pool of villages where biogas was installed, we first excluded those villages that were at a distance greater than 5 kilometers from the forest, as these are unlikely to be dependent on forests (Karnataka Forest Department, 2014). We divided the entire forest area into a 5-km grid, and identified controls as grid units where there were no villages with biogas interventions. We identified treatments as grid units where there were multiple villages with biogas interventions (proportion of households with biogas > 0.2, which was the lowest biogas adoption rate in the study area). Control and treatment grid units with comparable population density were adjacent to each other. This made the results directly comparable at the same population density (other predictors were highly correlated with population density, Table A in online Appendix). Because the forests were heterogeneous, and it was unclear how long the two blocks had been isolated, we chose to reduce the likelihood of species composition being very different at greater distances by selecting control and treatment grid units that were adjacent to each other (but see section 2.6).

2.5. Field surveys

Because the objective for field surveys was to assess forest recovery, we placed our sample in locations that could potentially be used by people but may not be used at present due to biogas intervention. Random location of plots within a

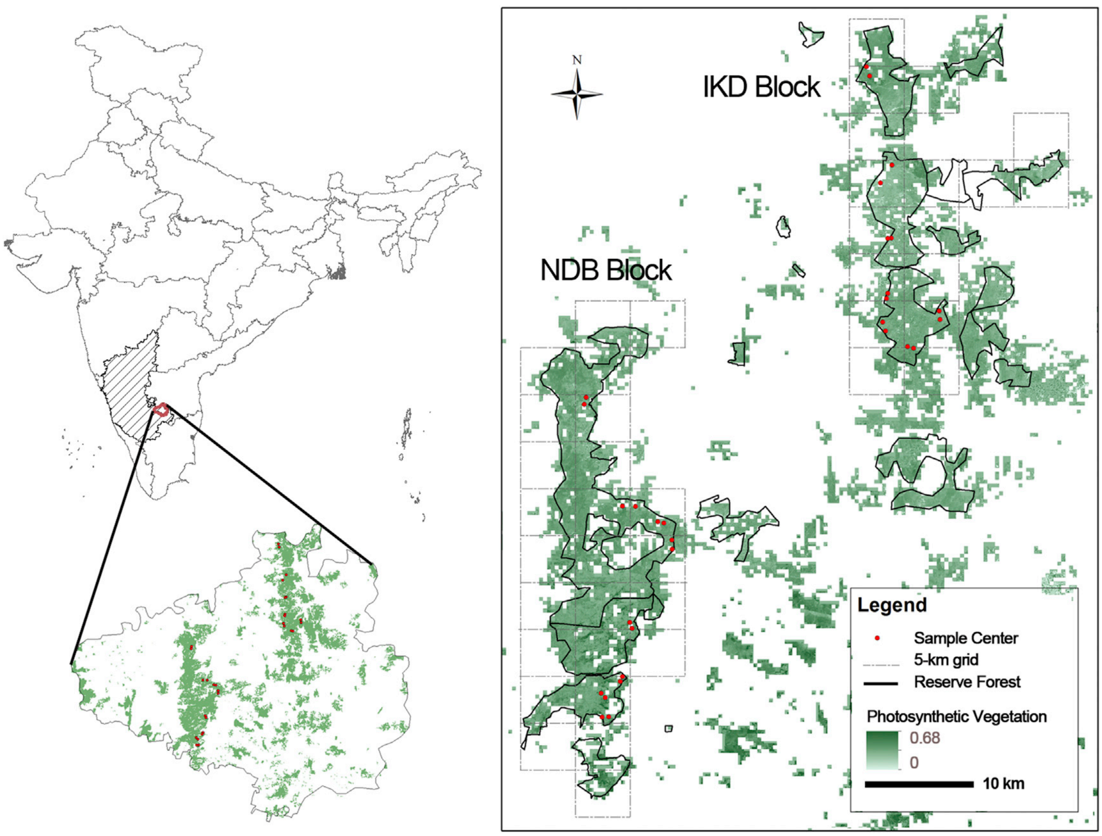

Fig. 1. Study area. Inset locates samples in the study area.

grid unit would not be informative due to differences in regular access of forests within the grid. People and cattle follow trails from their village into the state forest and minimize effort by preferring areas accessible from foot trails and those that have low slope. There were 2–4 trails from each village into the forest and people accessed forests close to the village even in grid units where intervention had taken place. Therefore, we placed our plots at an optimal distance on the trail where people would go and may have stopped. To select plot locations that were comparable in terms of accessibility, we walked along the paths used by the villagers and set our plots at a distance usually travelled by the villagers (distance obtained from asking 10 villagers in each village). To reduce variability due to slope, we also set plots in locations that had low slope in a 200 m × 200 m area. In locating our plots, we also excluded those areas with recent fire activity or Forest Department plantation activity (based on shapefiles provided by the Forest Department) as these would introduce variation in our system that we were not interested in examining. We used cardinal sampling design where we located 20 m × 20 m quadrats at 100-meter distance in four perpendicular directions from the center, and included one 20 m × 20 m quadrat at the center. To ensure that biophysical variables were not responsible for differences in species growth, we measured soil compaction using a Soil Compaction Tester (agraTronix, Streetsboro, Ohio, USA), and soil pH and nutrient levels by collecting soil samples at 0–15 cm depth (depths greater than 15 cm were not available) at two randomly selected points at each quadrat. These samples were analyzed for pH, electrical conductivity, organic carbon, phosphate, potassium, zinc, copper, iron, manganese, calcium, magnesium, sulphur and boron in the Agricultural Development Laboratory at Zuari Agrochemicals Limited. We also measured canopy cover using a spherical densiometer, and compiled information on temperature, precipitation, elevation, slope, and fire frequency for each site (Table 1). We also quantified fire scars in each plot. For quantifying tree population structure, we identified all plant and tree species in four size classes: large trees (>10 cm DBH), medium-sized trees (4–10 cm DBH), small trees (<4 cm DBH and height > 2.1 m), and saplings (height < 2.1 m), and we measured their DBH and heights. Within each subplot, we also used a 5 m quadrat to quantify and identify shrub species and seedlings. We used this data to calculate species richness and Shannon-Wiener Diversity Index, and used the information on DBH and height along with allometric equations to estimate aboveground biomass of trees (http://www.fao.org/docrep/w4095e/w4095e06.htm, Equation 3.2.2). We measured aboveground biomass of shrubs, saplings, and non-timber woody species using the reference unit method (Andrew et al., 1981; Kirmse and Norton, 1985; Lehmkuhl et al., 2013). We cut a small unit of the stem for each species and weighed it using a digital balance [0.01 g sensitivity]. We counted the number of such reference units that would fit into the plant, and estimated fresh weight of the plant by multiplying the number of reference units by the fresh weight of the reference unit. To obtain dry weight of the plant, we oven dried a subsample of the reference unit, and calculated the dry weight to fresh weight ratio and multiplied this by the estimated fresh weight of the plant.

2.6. Propensity score matching

Although our sampling design used pair-wise matching of grid units by placing controls and treatments in adjacent grid units with comparable population density, we imposed an independent match through propensity score matching (PSM) to ensure that the matching was independent and replicable (and we report results from both analyses). In a PSM, the first step is to calculate the probability of a sample being a treatment using variables that are collected prior to the analysis. The idea is to establish that there was no bias in selection of treatments and that treatments and controls are comparable. Because treatment villages were pre-selected at the time of biogas intervention in 2005 (see section 2.1 for details), we needed to first understand whether there were any differences in controls and treatments based on pre-existing factors. Therefore, we first estimated the probability of a village having a biogas intervention based on a set of variables that could be collected prior to the analysis (listed as biophysical, demographic and socio-economic predictors listed in Table 1). Because many of these variables were correlated (Table A in online Appendix), we used population density and forest block as predictors. We included forest block as a categorical predictor to ensure that matched samples were from the same block to increase the likelihood of similarity of species pool. We used a binary logistic regression where the response variable was the treatment (no biogas intervention = 0; biogas intervention = 1) to calculate propensity scores for each of our tentative plots (Table B1 in online Appendix). We used R package MatchIt to match samples (Ho et al., 2015), using both individual matching (method “nearest”) and propensity score-matched kernels (henceforth, Ps-matched kernels, using method “subclass”). For individual matching, the samples were perfectly balanced, which implies that controls and treatments were balanced and matched in our original pair-wise sampling design, and further analysis would only require a *t*-test between controls and treatments. Matching based on Ps-matched kernels led to a loss of six samples, leaving us with 22 samples in four sub-classes (two kernels could not be matched appropriately). Distribution of population pressure and forest block was balanced for controls and treatments in individual matching, and for the remaining subclasses for Ps-matched kernels (Table B2 in online Appendix). The distribution of propensity scores before and after matching shows that the variability is largely reduced and sample composition is balanced and appropriate for subsequent comparisons (Fig. C1 in Appendix). We saved the Ps-matched kernel identity and used it for further analysis.

We further needed to establish that the species compositions in treatments and controls were truly comparable, and that there were no differences in successional stages across matched treatments and control plots. We used Nonmetric Multidimensional Scaling (NMDS) ordination to test if there were significant differences in species composition between Ps-matched kernels for all species in a plot, as well as for species in different size classes. There were no significant differences in species composition between Ps-matched kernels for all trees (vegan package, adonis test, P = 0.12) (Oksanen et al., 2013), but species composition was significantly different in Ps-matched kernels for small trees (<4 cm DBH, adonis test, p-value 0.09) and medium-sized trees (between 4 and 10 cm DBH, adonis test, p-value 0.07) (Fig. C2 in Appendix C). This suggests that by testing within a Ps-matched kernel, we are controlling for differences in species composition.

2.7. Analysis

Our analyses used several parameters to assess forest recovery including vegetation structure, regeneration, reduction in invasive species such as *L. camara*, and relative abundance of plant functional traits (Ruiz-Jaen and Aide, 2005; Suding, 2011).

2.7.1. Structural differences

To examine whether biogas intervention was associated with differences in aboveground biomass, we used both our original pair-wise sampling design as well as PSM-matched samples. For pair-wise sampling design, we could use a one-tailed *t*-test to test if aboveground biomass was higher for sites where biogas was introduced. For PSM-matched samples, we compared aboveground biomass between control and treatment within a Ps-matched kernel by using the identity of the Ps-matched kernel as a random effect in a general linear mixed model (GLMM). Because the random effect accounts for the variation due to kernel identity, significantly higher biomass with treatment would tell us whether the intervention had a positive impact. Our model for this was:

$$Biomass \sim Treatment + Random\ \ Effect(Ps - matched\ \ Kernel) \tag{1}$$

To understand factors associated with differences in biomass (in addition to treatment), we used factors from Table 1 in generalized linear mixed models (GLMM) to identify factors associated with differences in aboveground biomass. We also used principal component analysis (PCA) on soil nutrients to generate principal components (PCs) to represent the variation in fewer dimensions (Table D in online Appendix) and used the PCs as factors in model selection. Our alternate models included those factors where between factor correlation coefficients were ≤ 0.3 (Table A in online Appendix) to avoid collinearity. Least AIC was used to select the best model (Burnham and Anderson, 2002).

2.7.2. Forest regeneration and reduction in invasive species

To understand whether the biogas intervention was associated with forest recovery, we also tested whether there were significant differences in abundance and biomass of invasive species (*L. camara* abundance), and abundance, biomass and species diversity of seedlings and saplings. We used *L. camara* abundance, and abundance and species diversity of seedlings and saplings separately as response variables in Equation (1) to test for significant differences between treatment and control plots.

2.7.3. Species composition

To understand changes in species composition, we first selected the thirty most abundant species in the study area (Table E1 in online Appendix). We used Equation (1) to test whether the relative abundance of each of these species was higher in treatment sites. We compiled information on root-suckers, fire-resistance, resistance to herbivory, shade tolerance, resistance to planting density and ability to thrive in poor soil conditions from systematic studies on these species (Troup, 1983) (Table E1 in online Appendix), and used Equation (1) to test whether the relative abundance of individuals with a specific trait was higher in treatment sites. Finally, we selected all species that were reportedly indigenous to the area (Karnataka Forest Department, 2014) for which we could find trait-based information (Table E2 in online Appendix). In order to understand whether there were differences in trait-based analysis for abundant species and indigenous species, we used Equation (1) to test whether the relative abundance of individuals of indigenous species with a specific trait was higher in treatment sites.

3. Results

3.1. Structural differences

Aboveground forest biomass in treatment plots was significantly higher than control plots for the entire study region (Fig. 2, one-tailed *t*-test, p-value, 0.049), but moderately significant for the individual blocks: IDK block (one tailed *t*-test, p-value = 0.081); and NDB block (one-tailed *t*-test, p-value = 0.056). Treatment was also moderately significant within a Ps-matched kernel (Fig. 3(a), GLMM, p-value 0.06), suggesting that presence of biogas intervention is associated with significantly higher biomass in the study region.

Yet, there was overlap in the confidence intervals because such natural experiments have high variation in the response due to biological effects. Forest regrowth was not uniform across Ps-matched kernels, and the effect of treatment appears to be higher with slope, distance to water, lower density of livestock and small ruminants, and lower incidence of fire (Fig. F in Appendix). The best model included treatment, sulphur content of the soil, slope, a principal component of soil nutrients that was associated with zinc, iron and magnesium (PC3), and livestock ($R^2 = 0.66$) (Table G.1 in online Appendix). However, only treatment, soil sulphur content, and slope were significant (Fig. 3(d), Table G.2 in online Appendix).

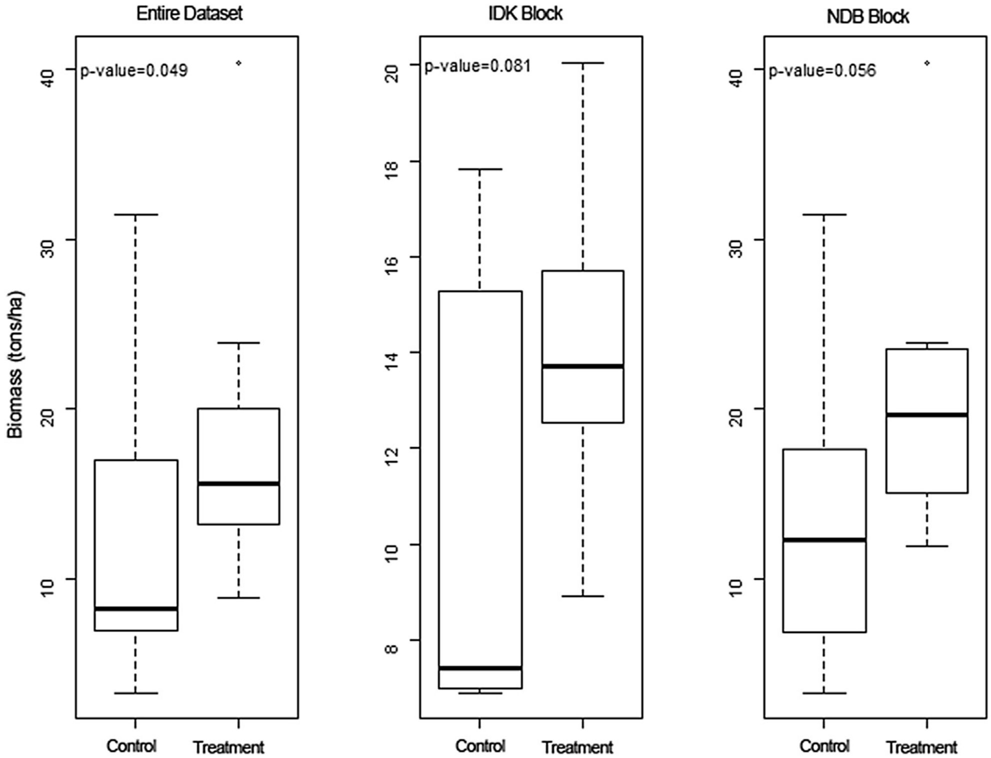

Fig. 2. Results from individual-based matching (n = 28) shows differences in biomass in control and treatment (where biogas interventions took place) in the entire study region, IDK block and NDB block. Boxplots represent mean and one standard deviation.

3.2. Forest regeneration and invasive species

Sapling abundance (GLMM, p-value 0.09) and sapling species diversity (GLMM, p-value 0.03) were significantly higher in treatment plots, even though there were no significant differences in seedling abundance (GLMM, p-value 0.29) and seedling species diversity (GLMM, p-value 0.68) (Fig. 3(b) and (c)). This result suggests that given a similar seedling diversity, individuals are more likely to reach sapling stage in treatment sites, but this may be attributed to differences in plant traits between control and treatment sites for seedlings (see section 3.3).

3.3. Species composition

Abundance of 13 of 30 species was significantly higher in treatment for some size class (seedling, sapling, small trees and all trees, Table H.1 in online Appendix). In general, there were similarities in traits which had significantly higher relative abundances across indigenous and abundant species (Fig. 4; details in Tables H.2 and H.3 in online Appendix). For species

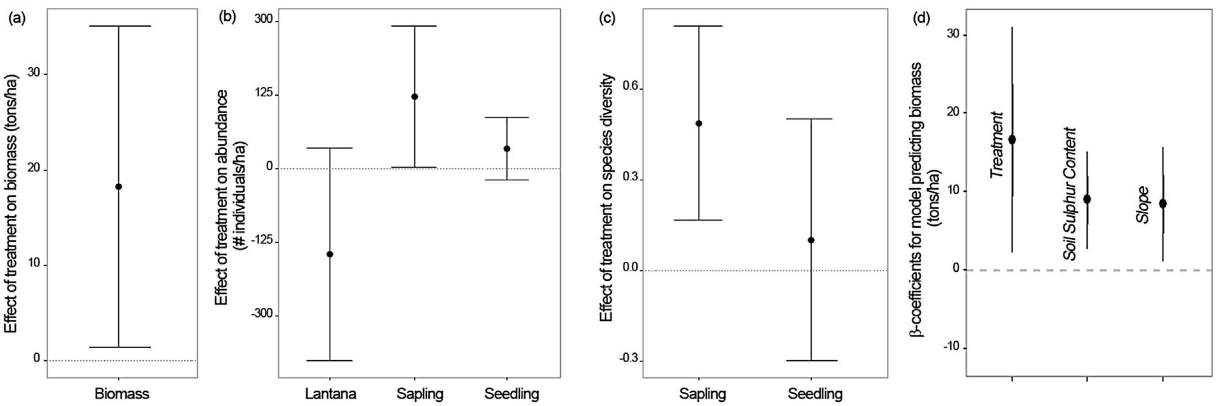

Fig. 3. Results from Ps-matched clusters (n = 22) shows estimated effect of treatment on (a) aboveground biomass, (b) abundance of *L. camara*, saplings and seedlings, (c) species diversity of saplings and seedlings, and (d) factors associated with differences in aboveground biomass (only significant variables shown).

composition, species that were not reported to be fire resistant when their traits were documented in detail (such as *Wrightia tinctoria*) had significantly lower relative abundance in treatment sites for Ps-matched kernels for all trees and small trees. Fire resistant seedlings were significantly higher in treatment sites for 30-most abundant species. Species that were light demanding and/or could not withstand shade as adults (such as *Anogeissus latifolia* and *Terminalia chebula*) also had higher relative abundance in matched treatments for seedlings, small trees and all trees. Finally, species that were not reported to do well in poor soil conditions (e.g. *W. tinctoria*) had significantly lower relative abundance in matched treatment sites for small trees and all trees. Small trees that suffer when density of individuals in the forest is high also had higher relative abundance in treatment sites for abundant species. Therefore, it is possible that regrowth in species of certain traits may be higher than other species.

4. Discussion

4.1. Measuring recovery

Forest recovery is often measured against a historical baseline or an undisturbed reference site, wherein similarity of vegetation structure, biomass and species composition with reference sites is an attribute of a restored system (Ruiz-Jaen and

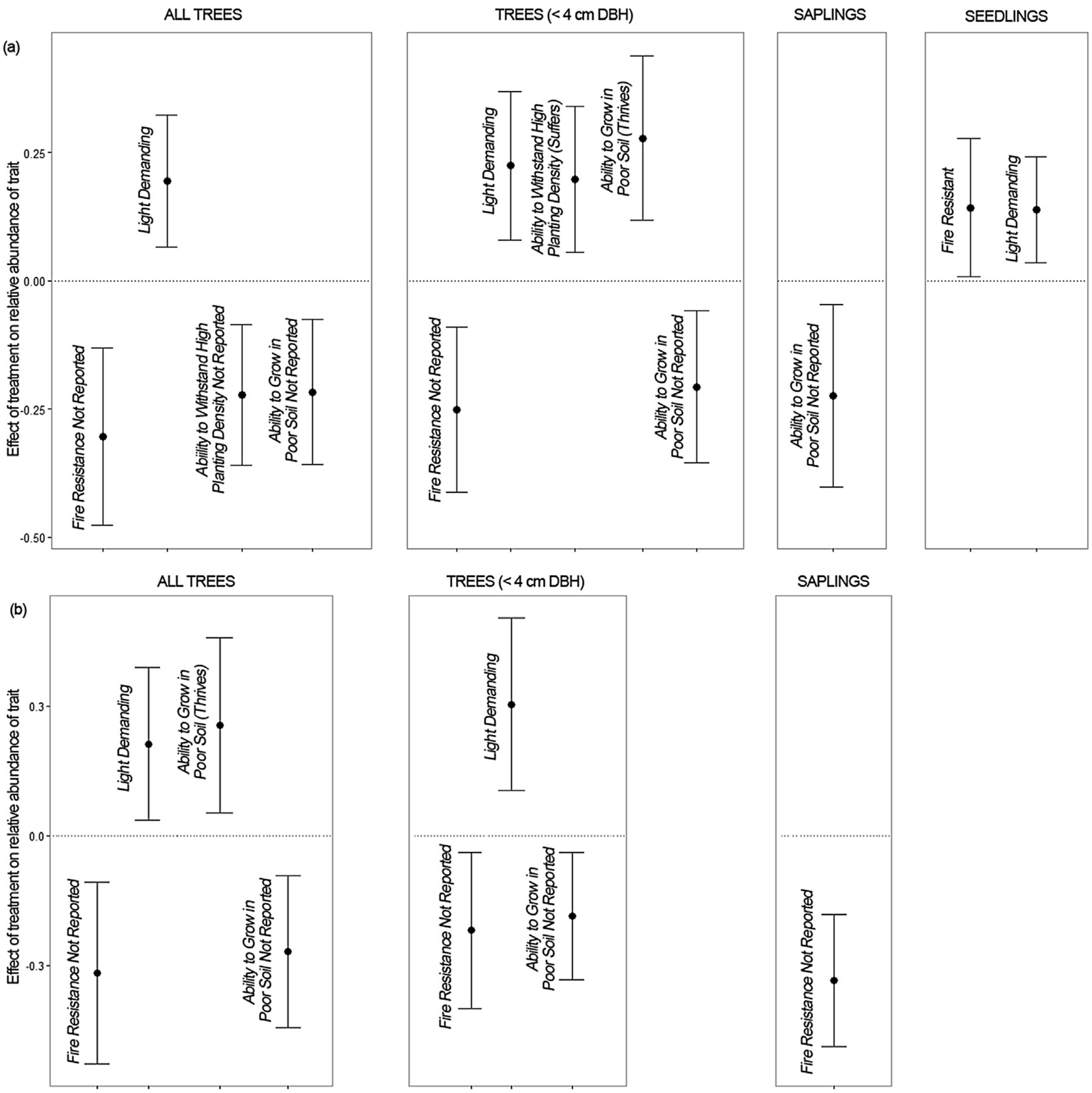

Fig. 4. Effect of treatment on relative abundance of individuals with certain species traits for (a) thirty most abundant species, and (b) indigenous species (only significant effects shown).

Aide, 2005). In the absence of such sites, recovery is compared against unrestored degraded sites but the parameters to be assessed become less clear – success is measured as increase in biomass, species richness, species diversity, colonization by a desired species, removal of non-native species, percentages of regional target species, and the trajectory towards reference conditions (Suding, 2011). Our study found that treatment (>20% of households with biogas) was significantly associated with higher forest biomass. These forests also showed long-term potential for regeneration as there was higher abundance and diversity of saplings despite there being no difference in abundance and diversity of seedlings. This result could be due to differences in plant traits for seedlings in control and treatment plots, where treatments had higher relative abundance of fire resistant and light demanding species.

4.2. Factors associated with recovery

Factors significantly associated with higher biomass include slope and soil sulphur content. Slope is often negatively associated with deforestation as people are less likely to use or convert higher slopes (e.g. Tucker, 1999), and the relative difficulty in accessing higher slopes on foot may explain higher biomass in areas with higher slope in our study area as well.

However, the mechanism for the influence of soil sulphur is less clear. Soil sulphur is an important plant nutrient, essential for parts of plant metabolism and physiology, particularly chlorophyll formation (Eaton, 1922; Webb et al., 2016). Sulphur is also often limiting in soils, and needs to be applied as a fertilizer (Eaton, 1922; Freney et al., 1962; Webb et al., 2016). Soil sulphur is found primarily in two forms: organic sulphur, whose proportion is associated with soil organic matter; and sulphates, which are associated with mineralization of organic sulphur and atmospheric deposition through rainwater (Eaton, 1922; Freney et al., 1962). But atmospheric deposition of sulphur is lower away from sources of industrial pollution and has reduced with reduction of coal-powered power plants (Eaton, 1922; Webb et al., 2016). Although soil sulphur content is associated with higher organic matter (Eaton, 1922), this is limited to organic sulphur, which is not the form in which plants utilize sulphur. Sulphates, the form in which plants use sulphur, are associated with mineralization from organic sulphur by fire (Freney et al., 1962; Gray and Dighton, 2006) and *Lantana* abundance (Osunkoya and Perrett, 2011). Due to the degraded nature of these forests, low quantity of manure or litter at some sites may have led to low quantities of soil sulphur, but low correlation of organic carbon and soil sulphur (correlation coefficient, $r = 0.24$) suggests that organic sulphur may be transformed to other forms—sulphates are mineralized by forest fires, and frequent forest fires may be responsible for increased sulphates in these forests (correlation coefficient between fire count and soil sulphur, $r = 0.50$). Further, because we had to rely on visual signs for fire estimation — MODIS was not able to capture the fires in the landscape — and factors such as rainfall and season may obscure fire signs, soil sulphur content may represent long-term fire activity better than visual signs and serve as a proxy for fire in this landscape. Studies have also found that *Lantana* abundance is associated with lower soil sulphur content, and high inverse correlation between *Lantana* abundance and soil sulphur suggests that this may occur in this site as well (correlation coefficient between soil sulphur and *Lantana* shrubs, $r = -0.35$). Therefore, soil sulphur limitation may be responsible for differences in forest regrowth, or it may just be a correlation and represent other factors such as high frequency of fires and low lantana abundance. Fire and *Lantana* abundance also show a clearer trend with forest regrowth (Appendix B), but causation needs to be established with detailed studies. Overall, abiotic conditions, landscape locations and past disturbance history can serve as barriers to recovery (Brudvig, 2011; Suding, 2011). This study suggests the same as abiotic factors such as slope and soil nutrients influence forest biomass.

Although success measured as recovery of biomass and species diversity is more common than recovery based on species composition (Suding, 2011), we were able to report change in relative abundance of functional traits for both abundant species and a list of desired species. This list was based on regional species pool (for dry deciduous and scrub forests in this region), historical information from the Forest Department, and availability of trait data. It is possible that trajectory towards regional target species or historical reference conditions are not definitive indicators of forest recovery because historical baselines may be shifting, arbitrary or unfeasible (Chazdon, 2008; Jones and Schmitz, 2009; Suding, 2011), but it is clear that relative abundance of species that are light demanding, fire-resistant and thrive in poor soil conditions (pioneer species) is higher in sites with biogas interventions. Increase in such traits may be expected in open forests with poor soil conditions and regular fires, but the relative increase in these traits in treatment sites suggests that extraction may have prevented growth in control sites. At the time of this study in the recovery trajectory, only ten years had passed and biomass recovers faster than species composition, and pioneer species recover before other species (Gignoux et al., 2016). It is possible that other species will also recover but this analysis is limited by sampling that occurred only ten years into recovery whereas recovery can take up to forty years (Chazdon, 2008; Jones and Schmitz, 2009).

4.3. Policy implications of biogas interventions

Although forest recovery and restoration is a global priority (Millenium Ecosystem Assessment, 2005), the proportion of interventions that have been examined is much lower than the number of interventions (Suding, 2011). These studies also focus on active and passive restoration from alternative land uses such as agriculture (Jones and Schmitz, 2009; Ruiz-Jaen and Aide, 2005) or from transformative extraction industries such as mining, deforestation, and logging (Jones and Schmitz, 2009). This study contributes to the growing literature on forest recovery by examining recovery from relatively low impact fuelwood extraction—an aspect of recovery that has not been examined before—and finds support for higher biomass and regeneration even if it is limited by site conditions such as soil nutrients and plant traits (light dependence, fire resistance and ability to grow in poor soil conditions).

This biogas intervention had very high on-ground support, and biogas units were in use and regularly maintained and repaired even ten years after the intervention first began. Such strong on-ground support suggests that the alternative fuel was provided consistently enough for communities to reduce their reliance on the forest, thus resulting in forest change. Biogas interventions with poor implementation or those in drought areas where it is difficult to support livestock may not have such results.

This study was conducted in degraded forests, and examples from other, less degraded sites may not show these limitations. Although many degraded sites require active restoration, passive restoration through reduced forest extraction may have the advantage of lower cost to benefit ratios and lower disturbance to existing ecosystems (Chazdon, 2008), in addition to its existing benefits for indoor pollution, diet and time allocation, and reduced emissions from burning for biogas interventions (Anderman et al., 2015; Feng et al., 2009; Garfí et al., 2012; Katuwal and Bohara, 2009).

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Appendix A. Supplementary data

Supplementary data related to this article can be found at http://dx.doi.org/10.1016/j.gecco.2017.06.005.

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