Anthropogenic drivers of headwater and riparian forest loss and degradation in a highly fragmented southern Amazonian landscape

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Word count: 8175 words

Abstract

Freshwater ecosystems across the Amazon are largely comprised of small streams and headwaters of riparian zones. These areas are legally protected within private landholdings in Brazil, but recent changes in the environmental legislation have slackened protection requirements, with unpredictable consequences to the integrity and functioning of these freshwater environments. Local drivers of riparian forest loss and degradation should be understood by considering context-dependent land management practices and pressures within a region. Here, we examine the spatial determinants of the total amount and spectral quality of both headwater and overall riparian forests within private landholdings within a highly fragmented region of southern Amazonia. We built generalized linear models to assess how the amount and spectral quality of headwater and riparian forests respond to landholding size and distance to roads and an urban center, and document landholder compliance rates according to both the current and previous Brazilian environmental legislation. Although forest loss and degradation are typically associated, forest degradation responded independently to the same drivers. Headwater forests were generally more degraded than riparian forests, and smallholders complied less often with legal requirements than largeholders. Proximity to roads and the nearest town had a detrimental effect on both headwater and riparian forest amount and quality, and distance to the nearest town affected all variables, except for headwater forest quality. Compliance with environmental legislation is the first step in protecting riparian and headwater forests, but alternative landscape management strategies must be explored, particularly focusing on smallholdings, which are most vulnerable to deforestation and forest degradation.

Keywords: compliance, deforestation, degradation, environmental legislation, private landholdings

Introduction

The Amazon basin is the largest tropical forest system on Earth, encompassing the largest and most complex hydrographic network. Amazonian freshwater ecosystems cover over 1 million km^2, drain ~6.9 million km^2 of moist tropical forests, and discharge 20% of the world's surface freshwater into the Atlantic Ocean (Coe et al. 2008). Over two thirds of the entire freshwater system in the Amazon is estimated to consist of small stream riparian zones (Freeman et al. 2007), including thousands of headwater regions of small drainage basins. This represents a pivotal interface between aquatic and terrestrial ecosystems, where water, nutrients, and sediments are exchanged (Godoy et al. 1999; Naiman et al. 2005). This hydrological system comprises an integral part of the Amazon basin, in that it provides key habitats for all aquatic and semi-aquatic biodiversity, regulates climate and water flow at local and regional scales, and promotes sediment and nutrient transport and cycling, among other ecosystem services (Naiman and Decamps 1997; Castello and Macedo 2016).

Areas of Permanent Protection (hereafter, APPs) are legally required set-asides prescribed by the Brazilian Forest Code (Brasil 2012), and include both riparian and stream headwater zones in addition to other fragile landscape features. Their primary goal is to maintain hydrological functions, although their role in maintaining other ecosystem services, such as soil stabilization and landscape connectivity for both terrestrial and aquatic biodiversity are also explicitly recognized in the legislation. In the Amazon region, riparian and headwater zones comprise the most ubiquitous forms of APPs. Recent politically motivated changes in Brazilian environmental legislation, however, have greatly reduced the legal requirements for the restoration of native vegetation set-asides within private landholdings, and granted legal amnesty for most non-compliant landholdings that experienced high rates of illegal deforestation in the past, despite vigorous criticism from the scientific community (Lewinsohn 2010; Metzger et al. 2010; Michalski et al. 2010b).

Most of the southern Brazilian Amazon has undergone severe deforestation since the late 1970s, creating an extensive fragmented landscape with varying degrees of forest cover (Michalski et al. 2008; Soler et al. 2009). This aging deforestation frontier was rapidly occupied by multiple waves of farmers in response to government-subsidized agrarian programs, and now largely consists of private landholdings of varying sizes. In contrast, state-managed protected areas are scarce (Michalski et al. 2010a), placing the fate of most forest ecosystems in the hands of private landowners. Most of the remaining natural vegetation cover throughout the tropics is now controlled by private and communal landholdings (Perfecto & Vandermeer 2008; Gardner et al. 2009), and over half of all natural vegetation in Brazil currently persists within the ~5.5

million private landholdings (Ferreira et al. 2012). Deforestation in this region is largely driven by the economics of agricultural and livestock enterprises, but the relative contribution of different actors — namely small and large landholders — depends on regional historical and socioeconomic contexts (Geist & Lambin 2002; Michalski et al. 2010a; Arias 2015). The discussion of how best to manage forest remnants or restore natural vegetation cover should therefore take into account context-dependent practices and pressures (Gardner et al. 2009). Understanding local imperatives of land stewardship that drive deforestation and forest degradation is critical in the discussion of how to counter-act the detrimental effects of policy changes, and should be done at the scale of individual properties, since this is the scale at which policy actions will ultimately be implemented (Aguiar et al. 2007; Gardner et al. 2013).

Tropical forest conservation science has largely focused on deforestation and fragmentation, whereas forest habitat degradation has been extensively overlooked (Ferreira et al. 2012). The Brazilian Forest Code (FC), which defines the minimum legal requirements for forest set-asides within private lands, is highly omissive concerning forest habitat quality. In this context, much work has focused on the contribution of small and large landholdings to forest loss (e.g. Aldrich et al. 2006; Aguiar et al. 2007; Michalski et al. 2010a; Gardner et al. 2013), while forest degradation has been widely neglected (Godar et al. 2014). Conservation actions planned under the UN/REDD+ (United Nations/Reducing Emissions from Deforestation and Forest Degradation) framework will require a high-resolution characterization of forest degradation patterns (Foley et al. 2007) and a more thorough mechanistic understanding of how both forest quality and forest amount are eroded over time (Gardner et al. 2012).

The goals of this study are therefore threefold. First, we describe the quantitative and qualitative patterns of riparian and stream headwater forests within APP areas in a highly fragmented region of southern Amazonia, at the scale of a ~900,000-ha municipal county. Second, we relate the amount of APP forests within 3,366 variable-sized private landholdings to identify the potential consequences of legislative changes to the FC compared to previous legal requirements. Finally, we assess the environmental, geographic, and land-tenure determinants of property-scale patterns of riparian forest integrity, and how these drivers affect the spatial distribution of these patterns across the landscape. We hypothesized that both the amount and quality of riparian forests would be affected by (1) landholding size, because largeholders control larger economies of scale, and are therefore better able to comply with the legislation compared to smallholders; (2) distance to urban centres, which is a proxy of the intensity of urban pressure exerted on forest patches; and (3) distance to primary and secondary roads, which is likely related to both the age of deforestation and the economics of exploitation of forest remnants.

Methods

Study area

The Alta Floresta county, located in the northern Amazonian state of Mato Grosso (09°53'S, 56°28'W), encompasses a highly altered landscape spanning 894,605 hectares, which has been severely deforested due to governmental incentives to establish bovine cattle farms in the region mainly during the 1980s and 1990s, following an ephemeral period of gold mining. Currently, a vast proportion of the county-scale landscape is comprised of cattle farms, forming a relatively homogenous matrix of low-quality exotic grass pastures in which forest fragments, riparian forests, and headwater forest patches of varying sizes and quality are embedded (Michalski et al. 2008). The Alta Floresta county contains a bovine herd size of 838,700 heads distributed across over 4,000 landholdings of varying sizes. The county now represents one of the most altered regions of southern Amazonian forests, an area known in Brazil as the 'arc of deforestation'. Alta Floresta is therefore ideal to study the effects of severe deforestation and degradation patterns, as well pinpoint potential management alternatives that can inform other regions of Brazilian Amazonia.

Landscape variables

The entire 894,605-ha study landscape was mapped using a supervised classification of 15-m resolution mosaic of RapidEye scenes, dated between July 2011 and August 2012. This classification was performed using the maximum-likelihood algorithm. We validated the resulting map using a χ^2 test of the confusion matrix using 243 independent ground-truthed GPS points (which were correctly classified more often than expected by chance: $\chi^2=200.97$; $p<0.001$). Overall accuracy obtained (number of correctly classified points / total number of validation points used) was 0.98. We were able to clearly distinguish five land-cover classes: (1) closed-canopy forest; (2) exotic grass pastures; (3) degraded and/or secondary forest; (4) low scrubby vegetation; and (5) and fast-growing tree (eucalyptus and teak) plantations. For the purposes of this analysis, we focused on the first three classes, because they were most prevalent across the landscape, and are associated with the process of large-scale deforestation and forest quality erosion, in which we were interested. Closed-canopy forest, cattle pastures and degraded/secondary forest comprised 46%, 45%, and 0.8% of the entire county area, respectively. Considering only riparian and headwater forest remnants, however, closed-canopy and secondary (or degraded) forest comprised 60% and 15% of the total area, respectively.

We obtained digital maps in vector format of the locations of all headwaters and streams/rivers across the study region from the Environmental Secretariat of Alta Floresta, following a detailed mapping assessment of the entire county, which were ground-truthed in situ (W. Butturi, pers. comm). The map was used to build a layer of 150-m buffer polygons (around points in the case of headwaters and lines in the case of streams), which we subsequently cross-referenced with our classified landscape map. These resulting maps were used to quantify the integrity status of forest patches, here defined as the total area of the three land-cover classes within the 150-m buffer around each headwater and riparian remnant (Fig. 1). The selection of this distance criterion allowed us to assess the degree to which remnant forest patches remained intact in a general context, since even past legal requirements sanctioned by the Brazilian FC are considered to be insufficient from an ecological perspective (Laurance & Gascon 1997; Lees & Peres 2008; De Fraga et al. 2011; Bueno et al. 2012). In addition, a larger buffer area would have been less sensitive to small-scale co-registration errors and any spatial incongruence between the shapefiles describing remnant forest patches and hydrological features. Stream vector lines had their origin in the headwater point locations, so that riparian buffers included headwater buffers. Therefore, in our subsequent analysis, we tested overall riparian integrity against headwater integrity alone.

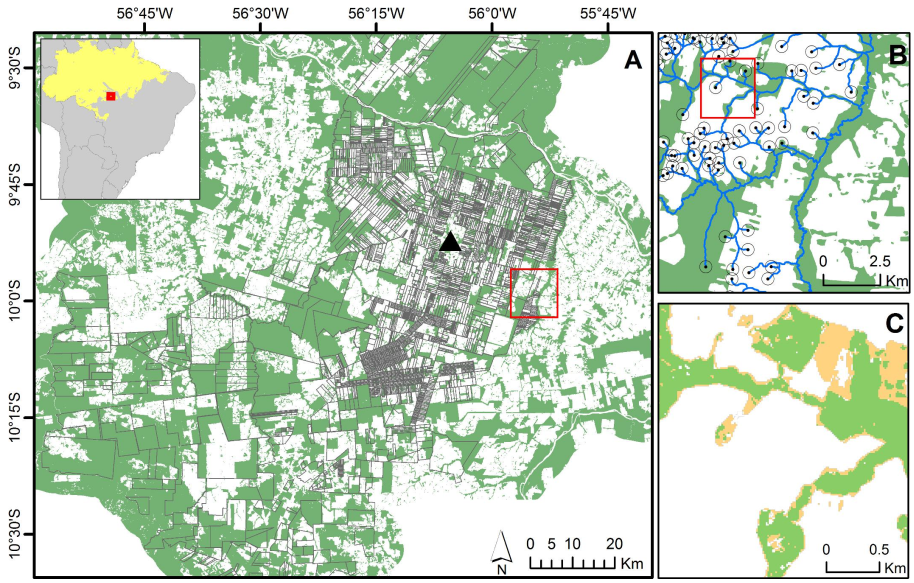

Figure 1. Map of (A) the Alta Floresta county, state of Mato Grosso, Brazil, corresponding to solid red square in inset map, showing the 3,366 private landholdings considered in this analysis (dark grey polygons); (B) example of headwater and riparian features within red square (in A); and (C) land cover classes (within red quare in B) obtained from the supervised classification of RapidEye images for both riparian and headwater forest remnants (closed-canopy forest in green, second-growth and degraded forest in light orange, and cattle pasture matrix in white). Solid triangle indicates the urban center of Alta Floresta. Inset map includes the phytogeographic contour of Amazonia in yellow.

A map of all main and secondary roads, both paved and unpaved, throughout Alta Floresta was obtained from Instituto Centro de Vida (ICV), a non-governmental organization based in the county. We also obtained a map of 3,366 private property polygons, which had been individually georeferenced across the entire county of Alta Floresta, from the Environmental Agency of Mato Grosso (SEMA), the municipal environmental agency, and private real estate companies. Landholding sizes ranged between 5 ha (in very small settlement plots) to 16,000 ha (in very large private landholdings), and encompassed a combined area of 65% of the entire county (Figure 1). Maps of headwater buffers and overall riparian buffers were cross-referenced with those of landholdings to calculate the status of forest integrity within each rural property. We considered the combination of all buffers (in terms of either riparian or headwater features) within individual landholdings as our response variable rather than individual buffers, since we were primarily interested in the persistence of forest areas within each landholding, and arbitrarily isolating discrete riparian forest patches is at best difficult. Therefore, in cases where a single landholding provided more than one independent buffer for headwater forests, for instance, these were combined and subsequently analyzed together.

One caveat in our analysis, however, is the land consolidation process that has taken place in some cases, in which several small landholdings can be purchased and ‘consolidated’ as a large landholding. In these cases, patterns of deforestation that could be attributed to a largeholder could in fact predate the larger coalesced landholding and most likely caused by a previous smallholder. In the few cases in which this was observed, often by detecting a conflict between databases, we decided to consider the older ‘unconsolidated’ property polygons in the analyses.

All remote sensing procedures were conducted in ENVI® v. 4.7, and all geospatial data processing was conducted in ArcGIS® v. 10.2.1.

Landholder compliance rates

We quantified the amount of closed-canopy and degraded forests within APP areas of each landholding, and calculated the property-scale compliance rate according to both the previous and the current legislation in the Brazilian Forest Code (FC). The previous FC legally required a 30-m wide forest strip to be set aside along each side of rivers and streams narrower than 10 m. The new legal requirements for these narrow watercourses, however, depend on the landholding size class as described in the current FC: Class 1 includes landholdings smaller than 100 ha, which are required to protect only 5 m of forest on either stream bank, regardless of stream width (represented by 2,661 of the total of 3,366 properties in our dataset); Class 2 includes landholdings between 100 and 200 ha, which are required to retain 8 m of forest on each stream bank (350 properties in our dataset); Class 3 includes landholdings between 200 and 400 ha, which are required to retain 10 m of forest on each bank (141 properties in our dataset); and Class 4 includes landholdings larger than 400 ha, which are required to protect 20 m of forest on either side of rivers and streams of any width (214 properties in our dataset). Wider rivers (width >10m) are legally associated with wider forest strip requirements, but since >90% of the drainage network consisting of 12,200 km of watercourses across the study area (equivalent to a hydrographic density of ~1.364 km of streams per km^2) is formed by narrow streams and most wider rivers were located outside our landholding map, we analyzed the integrity (amount and quality, described below) of these small streams only, and disregarded wider rivers in our study landscape.

For the protection of stream headwaters, the previous FC required a minimum 50-m radius forest patch to be set-aside around each headwater source point, but this has now been reduced to a 15-m buffer, representing a 91% area reduction in headwater protection, according to the new FC. Although the current FC prohibits any new deforestation above the thresholds defined in the previous FC, it effectively exonerated all deforestation violations that had occurred prior to July 2008 by waiving any forest restoration requirements. Because over 90% of all deforestation across our study region took place before mid-2008, this sweeping legal amnesty pardoning past forest clear-cutting becomes highly relevant in our study region and most of Brazilian Amazonia.

Data analysis

At the scale of the entire municipal county (894,605 hectares), we built generalized linear models (GLMs) to assess the influence of both local and landscape scale variables on the total amount of forest (proportion of both closed-canopy and degraded forest) and forest quality (proportion of closed-canopy forest only) within both headwater and overall riparian zones within each private landholding. Potential predictors of total forest amount included in the models were: (1) landholding size; (2) the straight-line distance to the Alta Floresta urban centre; and (3) the distance to the nearest main or secondary road. The variables we hypothesized to influence the quality of riparian and headwater forests included those described above, in addition to the proportion of forest cover within each landholding, which was used as a proxy of the size of remnant patches to account for the fact that smaller forest remnants experience disproportionately stronger edge effects compared to larger ones.

A total of four global models were therefore built, using a binomial distribution of the response variables. All independent variables were standardized and tested for collinearity. We did not detect high correlation values between the independent variables (Pearson's $r < 0.6$), so they were all retained in the analyses. All predictor variables were log-transformed to enhance linearity in their distributions. We detected a spatial autocorrelation in the global model's residuals (using Moran's I), and therefore performed a selection of spatial eigenvector filters to include these in subsequently re-analysed global models (Diniz-Filho & Bini 2005). The general fit of the global models was assessed by visual inspections of the residuals. We also tested for, but failed to detect, overdispersion in the data. We analysed models that contained all additive combinations of the hypothesized predictors, and performed a model selection procedure based on the Akaike Information Criterion (AIC) (Burnham and Anderson 2002). We used a model averaging procedure to extract the estimated beta-coefficients and confidence intervals for each variable.

All analyses were conducted in R 3.1.2 (R Development Team 2014), except for the selection of spatial filters, which were generated using SAM 4.0 (Rangel et al. 2010).

Results

We obtained information on 12,277 ha of headwater forests within 2,034 rural properties and 95,236 ha of riparian forests within 2,958 properties distributed across the Alta Floresta county. The vast majority of these properties are active livestock farms containing 73 ± 24% of mostly low-quality cattle pastures. Headwater forest retention was widely variable both in terms of total amount (proportion of any forest = 0.23 ± 0.28), and forest quality (proportion of CCF = 0.38 ± 0.37). A large number of headwater zones had been almost entirely cleared throughout the entire county (Fig. 2). Overall riparian forests, on the other hand, were often retained at greater extents (proportion of any forest = 0.59 ± 0.35), although they were also of highly variable quality (proportion of CCF = 0.33 ± 0.28; Fig. 3). Considering both forest amount and quality, the status of riparian and headwater forests (as described by the amount and quality of forest within the buffers) was particularly poor in the northeastern portion of the county, near the town of Alta Floresta, as well as in all areas dominated by smallholdings (Fig. 1 and 2).

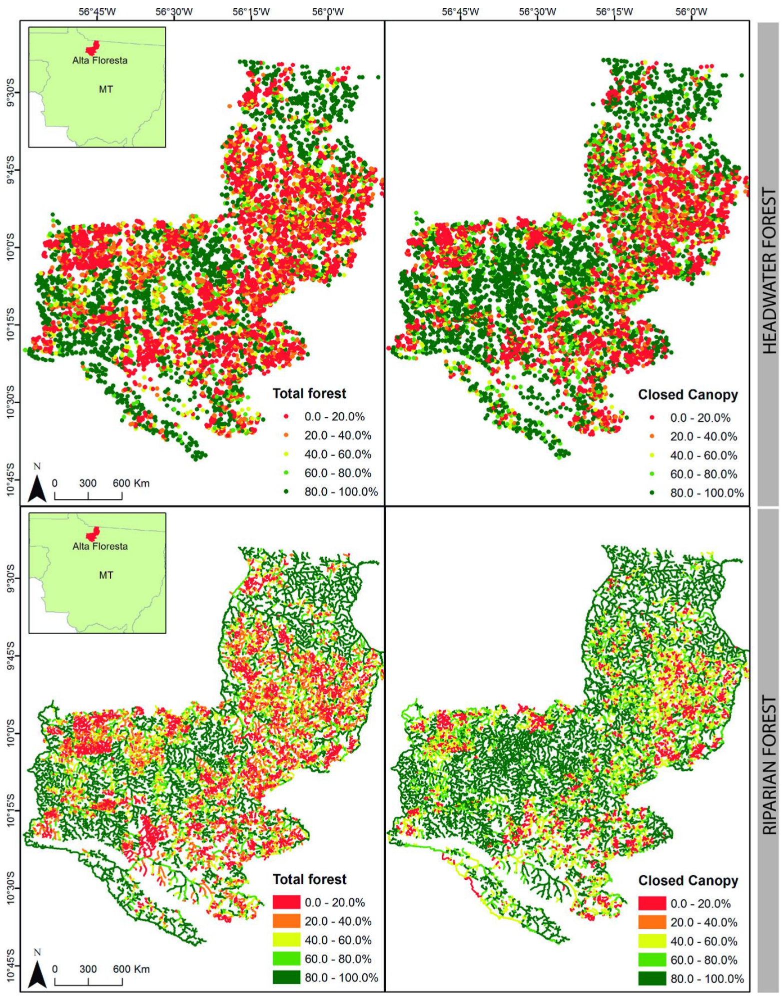

Figure 2. Heatmaps describing the general quantitative and qualitative status of headwater (upper panels) and riparian forests (lower panels) throughout the 894,605-ha Alta Floresta county, Mato Grosso, Brazil. Proportions of total forest amount (left panels) and quality, expressed in terms of closed-canopy forest (right panels), were colour-coded from lowest (dark red) to highest (dark green) based on 15-m resolution RapidEye images covering the entire county.

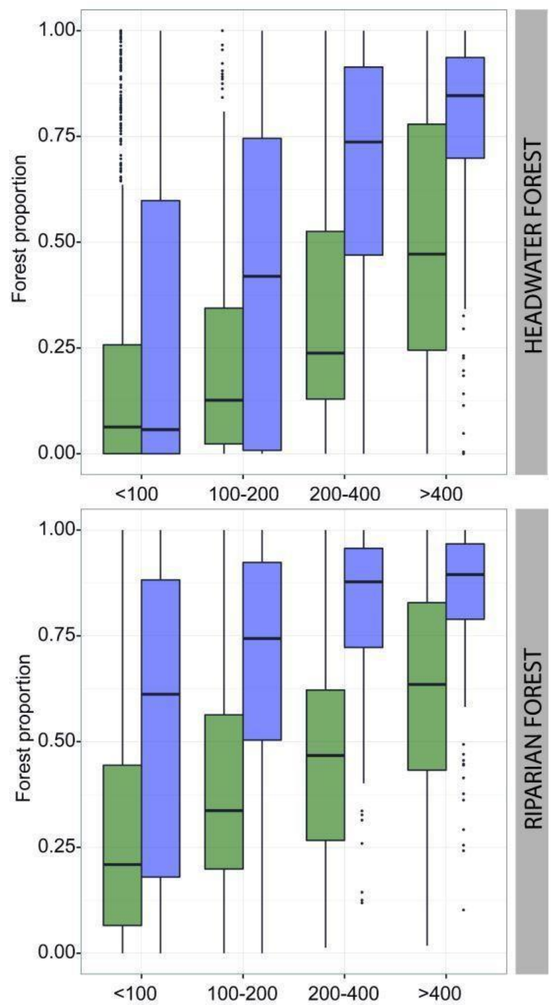

Figure 3. Boxplots show the median values of the proportion of forest (in green) and proportion of closed-canopy forest (in blue) retained within private landholdings as a function of landholding size class for both stream headwater (upper panel) and riparian forest areas (lower panel). Outliers indicate the large variation in the amount of forest retained for all landholding size classes.

Legally compliant landholdings that correctly set aside at least the minimum required headwater forest areas represented only 27.1% of all properties considering the previous 50-m requirement, and 44.8% considering the new 15-m requirement, but compliance rates varied widely with farm size (Table 1). Considering the requirements for each landholding size class in terms of riparian forests, we estimated that 84.9% of all landholdings are above minimum compliance levels with the new legislation. However, regardless of landholding size, compliance rates were much lower (58.2%) under the previous 30-m requirement, with smaller farm properties accounting for most of this deficit (Table 1).

Under the previous FC, there was an overall restoration deficit of 3,429 ha of headwater forests across the county, which could be attributed almost equally to both very small and very large landholdings. Class 1 smallholdings (<100 ha) accounted for 44% of the total area to be restored, while Class 4 largeholdings accounted for 40% of the deficit. Recent legislative changes will grant legal amnesty to almost 88% of those requirements. Riparian forests had a slightly lower landholding scale restoration deficit of 2,623 ha, but small landholdings (<100 ha) were responsible for 61% of this deficit. The recent changes in the legislation has accounted for a relaxation of over 2,389 ha (approximately 91%) in the overall restoration requirements, with smallholders benefiting most from these changes.

Table 1. Proportion of farms complying with both the previous and the current Forest Code (FC) legislation (both total landholdings and per landholding size class), considering the total amount of headwater and riparian forests maintained within a 150-m buffer.

	Total landholdings	Landholding size class (ha)			
		<100	100–200	200–400	>400
Headwater buffers					
Previous FC	27.1%	20.32%	26.99%	40.16%	67.16%
Current FC	44.8%	36.01%	46.37%	71.31%	88.56%
Total count	2034	1422	289	122	201
Riparian buffers					
Previous FC	58.2%	51.01%	74.85%	83.33%	94.12%
Current FC	84.9%	82.15%	95.03%	91.67%	94.12%
Total count	2958	2280	342	132	204

Forest remnants in both headwater and riparian zones were both proportionately larger and of higher quality in larger landholdings, with a stronger effect observed for riparian forests (Fig. 3, SM1). Both forest amount and quality, although positively associated with landholding size, were widely variable within each size class, indicating that much of the remaining forest is poorly protected even within large landholdings (Fig. 3). Both distance to town and to the nearest primary or secondary road were positively associated with the amount and quality of headwater forest (Fig. 4 and 5, SM1), which is consistent with our working hypotheses. Distance to roads was a significant predictor of both riparian forest amount and quality (Fig. 5), but distance to town was unrelated to the property-scale area of riparian forest retained (Fig. 4, SM1 and SM2). The overall proportion of forest within any given landholding also emerged as a significant positive predictor in the best models of forest quality (SM1 and SM2).

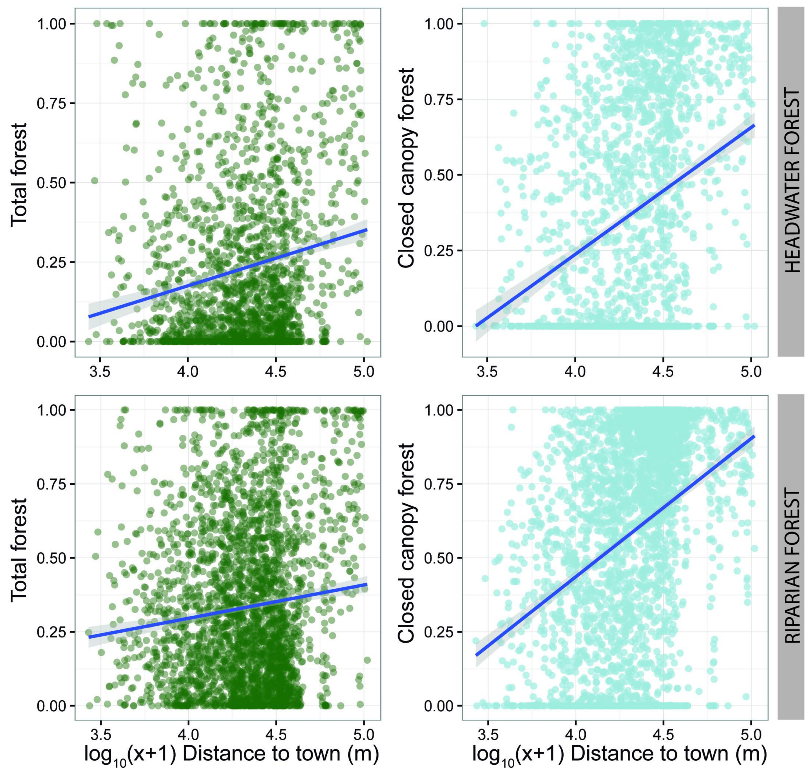

Figure 4. Distance (m) to the town of Alta Floresta positively affects forest quality (in blue) for both stream headwater sites (upper panels) and riparian forests along streams that were retained within private landholdings (lower panels), while affecting the proportional amount of forest (in green) for stream headwater sites only.

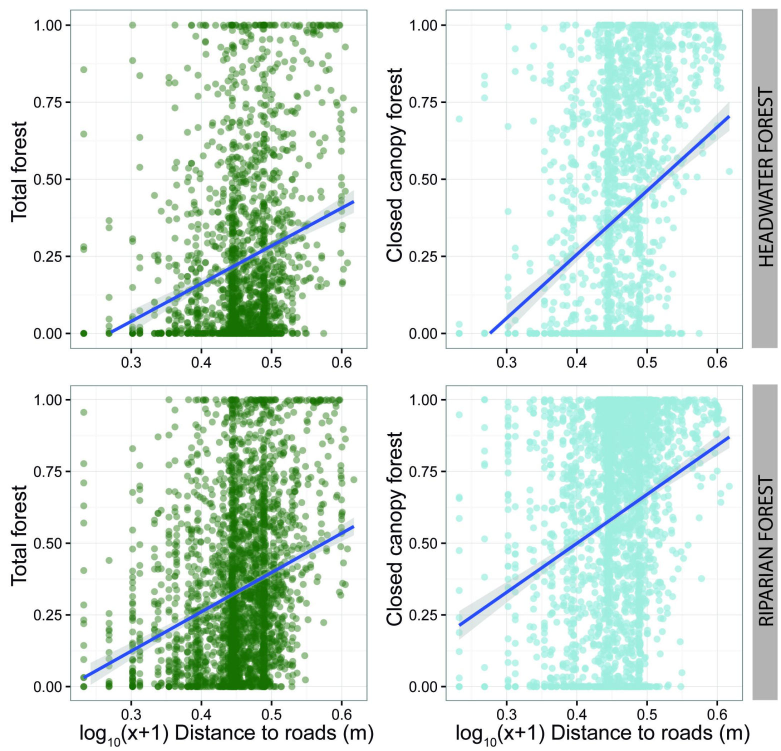

Figure 5. Distance (m) to primary and secondary roads positively affects the proportional amount of forest (in green) and forest quality (in blue) for both stream headwater sites (upper panels) and riparian forests along streams that were retained within private landholdings (lower panels).

Discussion

Conservation of freshwater ecosystems should consider the temporal and spatial connectivity that characterizes these systems, in which local effects of forest loss and degradation will likely have broader cumulative consequences (Castello et al. 2013). Temporal effects can be felt in changes in seasonal and supra-annual stream discharge and flood pulses, and spatial effects in changes in the longitudinal (upstream and downstream), lateral (between streams and adjacent land), and vertical (between streams and the atmosphere) exchange of water, nutrients, sediments, or organisms (Ward 1989, Pringle 2003), all of which can have synergistic and often unpredictable outcomes (Hayhoe et al. 2011, Neill et al. 2013). For instance, upstream deforestation, coupled with climate change, will alter surface water runoff, decrease evapotranspiration, and deplete below-ground water reserves, which have the potential to cause shifts in the water balance at local and regional scales, changing water fluxes and floodplain inundation patterns (Costa et al. 2009; Hayhoe et al. 2011). The loss of riparian forests will also add to these effects, as well as compromise the filtration of sediments and nutrients from land to stream, affecting water quality, and altering aquatic primary productivity (Williams et al. 1997; Neill et al. 2001). The immediate loss of protective forest cover along streams can increase water temperatures by up to 4°C, affecting water quality in terms of physical, chemical, and biological properties and changing the suitability of available habitats for a myriad of freshwater species (Macedo et al. 2013).

Riparian forests, therefore, act as buffers protecting stream geomorphology and water quality, mediating sediment and organic matter exchange, and providing access to clean water and other resources associated with perennial streams to many terrestrial species (Naiman & Decamps 1997). Although these riparian zones are used year-round, they become particularly critical as a water source during the dry season, which in the seasonally-dry southern Amazon is markedly pronounced. As we have shown for this study region (Zimbres et al. 2017), riparian forest strips are key ecological corridors for wildlife, promoting landscape-level connectivity, particularly under contexts of severe forest loss and degradation (see also Naiman et al. 1993; Lima & Gascon 1999; Uezu et al. 2005; Keuroghlian & Eaton 2008; Lees & Peres 2008; Martensen et al. 2008; Maltchik et al. 2008).

Curbing the loss and degradation of freshwater ecosystems requires high-resolution local information including (1) the location and amount of riparian forests, (2) riparian forest integrity, and (3) the drivers of loss and degradation (Castello et al. 2013). A quantitative assessment of the amount and habitat quality of riparian forests within private lands, including remnant patches around headwaters, is the first step in planning management strategies that can address local contexts and drivers, and identify priorities for protection and restoration efforts. The spatial distribution of riparian forest loss can be readily monitored through widely used techniques, such as a remote sensing approach (Castello & Macedo 2016). Forest degradation, however, is much less conspicuous, but can be detected in areas of naturally closed-canopy forest, such as in most of Amazonia. In a REDD+ framework aiming to mitigate both the effects of forest loss and degradation on national scale carbon emissions, both types of information are relevant (Nepstad et al. 2009).

Forest loss and degradation are typically associated, but the latter can also respond independently to the same drivers. In northern Mato Grosso, headwater forests were generally more degraded than riparian forests, and both large and small landholders cleared their headwater patches well beyond legal provisions, often removing them entirely. Headwater forest patch quality responded more strongly to landholding size than did overall forest amount, suggesting that largeholdings are able to retain more intact headwater forests regardless of the total amount of forest protected. Larger landholdings retained proportionally larger areas of riparian forests, which is consistent with other studies in the state of Mato Grosso (Oliveira-Filho & Metzger 2006; Michalski et al. 2010a; Caviglia-Harris et al. 2012). In other parts of the Brazilian Amazon, however, smallholdings can retain larger proportions of forest, especially in agrarian settlements established by the federal government (Godar et al. 2012; Godar et al. 2014; Medina & Godar 2016). This, however, may be due to the fact that subsistence agriculture comprises the main activity of these settlements, whereas even smallholders in our study region are effectively best described as commercial cattle ranchers.

We therefore found that smallholder compliance levels with environmental legislation was generally lower than that of largeholders, and this is almost certainly related to the much lower economies of scale of smallholders in meeting basic livelihood thresholds (Gardner et al. 2009; Michalski et al. 2010a, Peres and Schneider 2012). Although smallholders derive revenues from a smaller amount of land in absolute terms, they proportionally account for a higher deforestation footprint, in terms of their impact per landholding area (Fearnside 2005). This corroborates other studies suggesting that agrarian reform settlements should always be allocated to previously deforested areas, rather than to large redistributed forested landholdings (Fearnside, 2005, Peres and Schneider 2012). Smallholders also tend to exploit their remnant forest patches more intensively, for instance by harvesting higher basal areas of timber species or allowing cattle access to water and shade within forest remnants. Cattle intrusion into riparian forest strips is a major driver of forest degradation in small farms (Lees & Peres 2008). The cumulative effect of cattle intrusion on the density of understorey vegetation further corroborates the impact of cattle trampling and overgrazing on forest habitat quality and forest regeneration (Kauffman & Krueger 1984). This phenomenon may also take place in large landholdings, but cattle access to riparian zones therein is typically concentrated on a few sites, rather than the entire length of the riparian strip. Clearing pastures from invasive vegetation using fire is another widespread and inexpensive management technique used in the past by smallholders. Uncontrolled wildfires, however, often intruded into remnant riparian forest strips, thereby adding to cumulative degradation within many of these remnants.

Road networks have been shown to impact forest-dependent biodiversity at larger scales (Aguiar et al. 2007; Moura et al. 2014), whereas we here demonstrate that this effect holds at a finer spatial resolution, since both the amount and quality of headwater and riparian forests were greater in patches far removed from primary and secondary roads. Roads are often related to cryptic patterns of human disturbance, which are facilitated by greater access to remnants (Ahmed et al. 2014). At regional scales, roads represent easier access to otherwise isolated forest tracts and pave the way for direct drivers of forest disturbance, including timber extraction, wildfires, and hunting (Peres et al. 2006). At more local scales, roads are associated with the age structure of landholdings and the intensity of disturbance. In Alta Floresta, older rural properties were first established near the main primary and secondary roads, whereas more recently demarcated properties are more remote and accessed by smaller privately managed roads, many of which are not officially mapped. The county's official road network therefore serves as a proxy for the timing when deforestation took place. Landholding age is an important determinant of the amount of forest retention and the type of management practiced within forest remnants (Pfaff 1999). Michalski et al. (2010a) noted that more recently established properties in the region of Alta Floresta retained larger amounts of riparian forest, and this pattern holds true even in small landholdings. Although many of the oldest landholdings, which had been largely deforested early in the colonization history of Alta Floresta, may have partly regenerated, we hypothesize that old properties have in most cases experienced more severe patterns of forest degradation, since the cumulative exploitation of timber resources in forest remnants was and still is common practice throughout the region. Secondly, the degree of access by major roads can also determine the history and intensity of mechanized logging due to logistic restrictions in accessing remote forests with heavy equipment.

Distance to the urban centre affected riparian forest quality, but not the proportional retention of riparian forests. In our experience working in the study region since 2001, urban stressors that may affect forest habitat quality include occasional timber extraction, and wildfires accidentally generated by fishers and hunters. Illegally discarded trash is also a common occurrence in peri-urban riparian remnants, and many patches were consequently littered with trash. Even though the Alta Floresta town contains fewer than 50,000 inhabitants, the amount of pressure exerted on neighbouring forest remnants is extensive. In contrast, both headwater forest amount and quality were affected by distance to town, which corroborates our observations of negligent landowners in their attitudes towards headwater forest patches.

The fact that both riparian forest clearing and degradation responded synergistically to the same drivers in similar ways indicates that coordinated actions can address the protection of both forest amount and quality in the Alta Floresta region. A pragmatic understanding of the specific context and drivers of anthropogenic impacts includes identifying why smallholders clear and degrade their remnant patches (Arias 2015), while recognizing that there are fundamental societal trade-offs between agricultural land use and conservation land sparing (DeFries et al. 2004; Gardner et al. 2009). The implementation of management strategies will ultimately take place at the scale of municipal counties, so that strategies that best serve the reality of individual counties should be identified (Gardner et al. 2013). A series of conservation efforts in the Amazon since the early 2000s have contributed to a decrease in overall deforestation, including tightening of law enforcement, restrictions on access to rural credit by deforesters, and interventions in the supply chains of soy and beef from municipal counties showing high illegal deforestation rates (Nepstad et al. 2014). In the Brazilian 'arc of deforestation', counties such as Alta Floresta that are more accessible to state capitals and other parts of Brazil are more heavily subjected to law enforcement actions, and more policy-responsive (Godar et al. 2014). Indeed, past actions to curb deforestation, in order to remove Alta Floresta from the 'Red List' of deforestation hotspot counties, have demonstrated that coordinated management efforts can succeed in promoting forest preservation and restoration. In this case, most changes have been observed within medium to large landholdings (Coudel et al. 2012), which are more responsive to law enforcement and restrictions on credit access (Richards and VanWey 2016).

However, safeguarding compliance with environmental legislation is only a first step in promoting forest protection and restoration, while even under the very lenient set of requirements prescribed by the new FC, compliance rates are still low, particularly for headwater forests. In Brazil, minimum restoration requirements for native vegetation were reduced by 58% after changes in environmental legislation (Vieira et al. 2014). In our study landscape, minimum restoration requirements declined by 88% for headwaters and 91% for riparian forests, even though the previous deficit was not very high across the county (3,429 ha and 2,389 ha, respectively). Current legislation, however, is entirely mute in relation to forest quality, since the existing FC makes next to no requirement concerning the status of forests to be set-aside or restored, stating that native vegetation within APPs can be either primary or secondary at any stage of regeneration (Brasil 2012). Landowners can therefore inadvertently drive a process of widespread forest degradation even under scenarios of full law compliance. Also, recent rebounds in deforestation and forest degradation rates, following the Forest Code legislative reform, indicate that any successful outcome of environmental policies can be ephemeral and vulnerable to both market pressures and political expediency (Castello & Macedo 2016). Continuous vigilance and tightening of law enforcement, and policy pressure on market access affecting largeholder revenue are still crucial to keep them on track, since in absolute terms they control the largest remnant forest areas within their properties. For example, in our dataset large properties contained over 70% of all forest remnants even though small farms accounted for nearly 80% of all landholdings.

On the other hand, alternative strategies engaging local landholders in protecting forest reserves beyond the legal requirements must be considered. Additionally, applying sanctions to smallholders, who are often poorer, can be socially unacceptable and illegitimate, thereby creating conflicts (Godar et al. 2014; Arias 2015). In Alta Floresta, suicide rates among smallholders have increased over the last decade (W. Butturi, pers. comm.), and this is undoubtedly associated with prohibitive environmental compliance targets. Small landholders are governed by a different set of motivations and behaviours, but the much lower compliance rates observed for this landholding size class could be related to a lack of access to information, credit, and technical support (Gardner et al. 2013; Arias 2015; Nunes et al. 2015), rather than a negative attitude towards forest preservation (Coudel et al. 2012). In the state of Pará, landowners complained that they lacked technical guidance in engaging in management activities, and that this posed a greater obstacle than issues of credit access (Vieira et al. 2014). In fact, smallholders often choose to retain forest reserves within their land, but report that they are forced to clear remnant forest patches to sustain a minimum level of production for their families (D'Antona et al. 2006). Smaller landholders may therefore respond better to monetary incentives such as PES (payment for ecosystem services) schemes, which are still incipient in the Brazilian Amazon (Coudel et al. 2012; Garcia et al. 2013; Peres et al. 2013).

In Alta Floresta, recent incentives promoted by a municipal project for fencing and restoring headwater patches and riparian strips have been implemented, in which monetary and technical support prioritized small landholders (Secretaria Municipal de Meio Ambiente 2014). Fencing of riparian forests prevents overgrazing and trampling of vegetation by cattle, clearly the most widespread source of forest disturbance and degradation, and allows for the natural regeneration (or active restoration wherever it may be the case) of the forest. In this project, the municipal government distributed all materials necessary for building fences around degraded headwater and riparian remnants within smallholdings, and engaged them into active restoration actions. Such strategies should be assessed and implemented elsewhere in the Amazon, particularly in cattle ranching regions, since there are several benefits of such smallholder engagement. For instance, the municipal project additionally provided smallholders with technical support to register their properties in the Rural Environmental Registry, which is required by law but is often neglected by smallholders due to costs of georeferencing their properties and lack of know-how in using a digital platform (Secretaria Municipal de Meio Ambiente 2014). Such Registry, prescribed in the Forest Code, will become the main tool to monitor compliance and mitigate forest loss and degradation regionally, since it makes information on the geographic location and boundaries of farm polygons, and their forest remnants and APPs widely available.

Therefore, as a post-hoc analysis, we calculated the total length of fences that would be required to isolate all headwater and riparian remnants, based on real local transaction costs for fence-building in Alta Floresta, including wires, fence posts, and labour (see Table 2). A total of R$61,180,294 (≈ US$19,735,579 at an exchange rate of 3.10) would be necessary to fence all 1,109 riparian and headwater forest patches within all landholdings considered in this study (Table 2). Excluding cattle from forest patches within small landholdings only (<400 ha) would cost R$24,085,995 (≈ US$7,769,676) overall. The average cost for a smallholder would be around R$25,656 (US$8,276), although these costs range widely across smallholdings depending on the size of forest remnants (R$4,050 – 248,716, or US$1,306 – 80,231). These costs are obviously prohibitive to smallholders, and providing the technical support and monetary incentives to promote these measures would be highly beneficial, particularly in municipal counties containing a large proportion of smallholdings and agrarian reform settlements.

Table 2. Mean [range] length of fences and monetary cost of fence-building to exclude bovine cattle from all riparian and headwater forest remnants for each landholding class across the Alta

Floresta municipal county, based on real transaction costs informed by local landholders. Total length of fences and costs scaled to the entire county are also presented.

Landholding size class (ha)	Number of landholdings	Total length (m) of fences	Fencing costs (US$)[1]
<100	675	1,342 [300–7,940]	5,844 [1,306–34,575]
100-200	187	2,653 [318–12,162]	11,552 [1,385–52,965]
200-400	77	4,965 [359–18,423]	21,622 [1,561–80,231]
>400	170	16,163 [451–13,004]	70,388 [1,963–566,326]
Total	1,109	4,531,873	19,735,579

[1] These costs are corrected for inflation (as of January 2017) based on a 27.68% correction factor.

We finally highlight the fact that the response variables we examine here are proxies of real factors driving forest integrity. Our models would have been improved and the underlying processes of forest loss and degradation better understood if specific direct drivers had been considered. These include access to hunting or leisure activities by landowners, presence of dams within riparian strips, the true age of properties, whether or not a landholding is a family residence, and the history of fire use and timber extraction, to name a few. In any case, proxies are useful in that they can ensure a large-scale assessment and point to specific directions where local actions should be implemented.

Conclusion

Riparian APPs are the best available opportunity in Brazil and many other tropical countries to consolidate landscape-scale connectivity networks that would safeguard key hydrological functions of the land-water interface of freshwater ecosystems as well as provide dispersal corridors for biodiversity (Peres et al. 2010, Lees & Peres 2008, Zimbres et al. 2017). Beyond compliance with environmental legislation, the identification of strategic sites under pressure is important to help focus conservation priorities, and promote the implementation of such measures. There are no legal tools that explicitly require such planning, and efforts will need to be fostered by other means. Successful actions in the past have shown that curbing deforestation can be accomplished, but the restoration of forest habitat quality must also be included in the discussion, since the limited value of low-quality remnants for connectivity has already been demonstrated (Harrison 1992; Bennett et al. 1994; Lees & Peres 2008). Ensuring that landholders are able to comply with the legislation is therefore just the initial planning stage for effective conservation, and in highly fragmented landscapes presenting a large proportion of smallholdings, alternative ‘carrot and stick’ motivational strategies must be explored.

Acknowledgments

We are grateful to Weslei Butturi of the Secretaria Municipal do Meio Ambiente de Alta Floresta for sharing GIS files with us. The Brazilian Ministry of Education (CAPES) funded BZ’s PhD studentship. We thank the University of East Anglia for hosting BZ during a study visit. CNPq provided a research grant (#306392/2013-5) to RBM. We also thank IdeaWild Organization, Rufford Small Grants Foundation (#12658-1), National Geographic Society/Waitt Grant (#W314-14), and a CAPES grant to CAP (004-2012) for financial support for the fieldwork in Mato Grosso, Brazil. We are indebted to all landowners of northern Mato Grosso for granting access to their properties.

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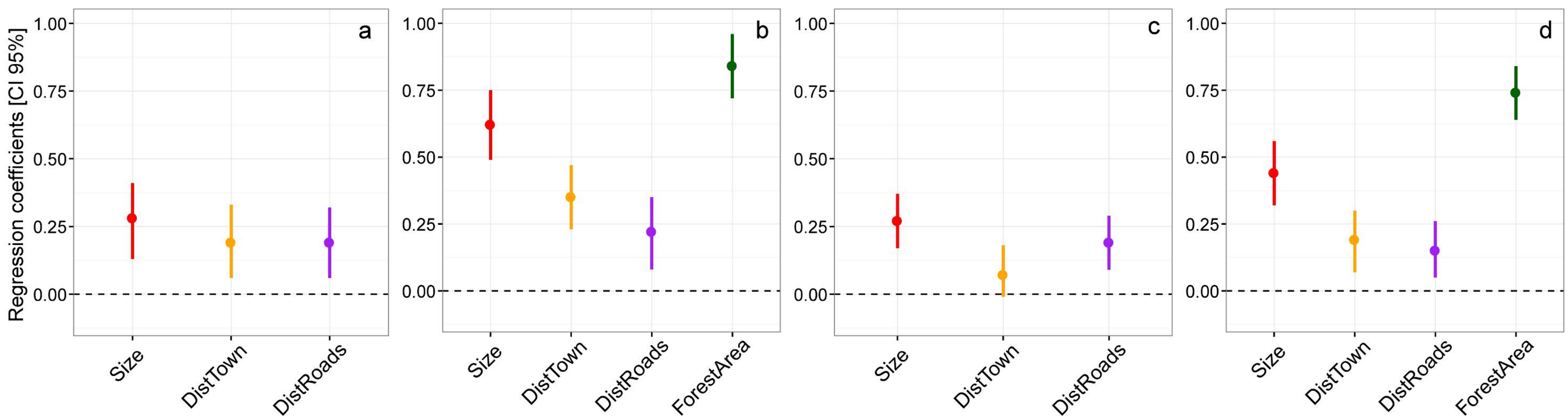
Supplementary Material



SM1. Regression coefficients (and 95% confidence intervals) for all variables included in the global models, and obtained by model averaging. Models were generated for (a) amount of headwater forest; (b) quality of headwater forest; (c) amount of riparian forest; and (d) quality of riparian forest.

SM2. Model selection results based on Generalized Linear Models (GLMs) with ΔAIC < 7, for the assessment of regional explanatory variables of total forest amount (within a 150-m buffer) and forest quality (proportion of closed-canopy forest) for both headwater and riparian forests within private landholdings. Int = intercept; Size = landholding size (ha); Dist town = distance to town (m); Dist roads = distance to primary and secondary roads (m); FA = total forest area within landholding (ha).

		Int	Size[a]	Dist_town[a]	Dist roads[a]	FA	LogLik	AICc	ΔAIC	wAIC
Headwater forests	*Amount*									
	Model 1	-1.26	0.28	0.19	0.19	[b]	-896.50	1803.0	0.00	0.92
	Model 2	-1.26	0.32	0.13	0.20	[b]	-900.58	1809.2	6.14	0.04
	Quality									
	Model 1	-0.60	0.62	0.35	0.22	0.83	-987.91	1987.9	0.00	0.99
Riparian forests	*Amount*									
	Model 1	-0.73	0.27	0.08	0.19	[b]	-1624.88	3259.8	0.00	0.88
	Model 2	-0.73	0.29	–	0.20	[b]	-1628.03	3264.1	4.30	0.10
	Quality									
	Model 1	0.49	0.44	0.19	0.15	0.74	-1443.06	2900.2	0.00	0.99

[a]Variables were log-transformed.
[b]Variable that were not included in the model.