

THE GROWING DILEMMA OF TIMBER HARVESTING IN BRAZIL NUT RICH,  
COMMUNITY FORESTS IN NORTHERN BOLIVIA: EFFECTS ON NATURAL  
REGENERATION AND FOREST DISTURBANCE

By

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To Cornelia, to whom I owe what I am  
Este logro es también tuyo Madre

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Desire for economic returns is stimulating increased logging in communally-owned Brazil nut (*Bertholletia excelsa*)-rich forests in Northern Bolivia. Building on previous research showing low logging damage to mature *B. excelsa* trees, we compared the effects of formal and informal logging (with and without management plans, respectively) on natural regeneration (individuals  $\leq 10$  cm dbh) of two non-timber species, *B. excelsa* and *Hevea brasiliensis*, and 10 timber species. Logging intensity was no different between formal and informal logging (0.28 vs 0.24 trees ha<sup>-1</sup>;  $p = 0.378$ ). Contrary to our expectations and reported industrial-scale findings, formal logging in these communities resulted in a larger percentage of disturbed area than informal logging (10.6%  $\pm$  0.65 SE vs 6.9%  $\pm$  1.26 SE;  $p = 0.012$ ). This may be attributed to the fewer trees extracted in informal logging, and to the often-ignored planning community landholders conduct with informal loggers prior to timber extraction. Regeneration densities of *B. excelsa* did not differ between unlogged or formally- or informally-logged sites measured 2–5 years after logging. *B. excelsa* densities were, however, greater in larger logging disturbances (i.e., log landings vs. skid trails), corroborating our model in which canopy openness explained regeneration densities ( $p = 0.001$ ). Regeneration responses of timber species varied by logging

treatment and type of disturbance, but all species demonstrated higher densities in the core zone (versus edge zone) of disturbed areas. Regeneration of two of the most commercially important timber species (*Swietenia macrophylla* and *Amburana cearensis*) rarely occurred, and only where conspecific trees were felled, implying that population recovery of select species may only occur with silvicultural interventions.

Findings that neither formal nor informal logging affected regeneration densities of *B. excelsa* within communally logged forests suggest that multiple-use—Brazil nut-timber—may be viable under certain circumstances. Timber harvesting at low intensities as in our study sites (0.13–0.31 trees ha<sup>-1</sup>), and integrating local knowledge and practices, such as opening short paths to reach target trees as practiced by community landholders, are two observed management features that may facilitate more sustainable multiple forest use. Integrating and adapting these and other practices, policies and support mechanisms that promote small- versus large-scale logging activities among communities could help bring traditional forest users into the legal framework while also conserving biodiversity and reducing deforestation and forest degradation.

## CHAPTER 1 INTRODUCTION

The creation of both the land reform and the forestry laws in the late 1990's resulted in substantial changes in Bolivia's rural social context and forestry system (Benneker, 2006). Most notably, it granted legal rights to forest dependent communities and included informal loggers into the legal framework. Although secure land tenure itself has the potential to encourage sustainable community forest management (Benneker, 2006; García-Fernández et al., 2008), whether these policies facilitate long-term sustainability of community forests remain unknown. Meanwhile, informal logging is spreading at the same pace as timber market diversification, and so are the ways in which informal logging occurs. Despite the struggles of implementing the forestry law and dealing with increased illegal logging in communally owned lands, the combined area managed by indigenous and smallholders has risen to almost equal that of medium- and large-scale landowners (1.25 and 1.52 millions of hectares, respectively) (Superintendencia Forestal, 2007). Most of these managed areas have had external support and/or assistance from local governmental institutions or national and international nongovernmental organizations (NGOs). Still, support has not reached the majority of communities, particularly the smallest ones, which usually have to rely on third-parties to carry out most logging activities taking place in their forests. Although these third-party loggers often comply with the basic rules of the forestry law within community lands, they rarely apply RIL techniques to prevent unnecessary damage and/or improve natural regeneration of commercially important species. Third-parties perceive community owned forests as providers of immediate economic benefits; therefore, they tend to ignore the importance of the forest for its long-term productivity, provision of ecosystem services and biodiversity value. Despite the socioeconomic losses illegal logging represents to governments (Kaimowitz, 2007), informal logging can be up

to five times more profitable than formal logging (Rice et al., 1997); thus, the ability to control forest activities weakens as illegal logging becomes increasingly widespread (Kaimowitz, 2007) and economically more interesting than formal logging (Rice et al., 1997).

To guarantee long-term timber yield, Bolivia's forestry law largely relies upon generalized simulations on species growth and recovery rates (Dauber et al., 2005). In a wide range study of distribution patterns of tree species along Bolivian lowland forests, Toledo (2010) has shown, however, that natural regeneration of each individual species responds uniquely to logging. Therefore, species specific forest management recommendations would enhance current generalized management guidelines. Another important measure to consider is the extent of disturbance caused by logging, which ranges between 5 and 50% of the logged area in tropical forests, varying with harvest intensity, yarding method, and the care logging operators take to reduce logging impacts (Putz et al., 2001). For example, increased logging intensity may cause significant mortality of advanced regeneration (Gerwing, 2002). Furthermore, logging high volumes of conspecific species would typically compromise regeneration of young individuals, because most tropical tree species are found in low densities (Gerwing, 2002). However, felling gaps from logging may have positive implications for natural regeneration, especially in the establishment of high value species (Gullison and Hardner, 1993; Johns, 1996). Despite significant gains in our understanding of logging effects in tropical forests (Johns, 1996; Pinard and Putz, 1996; Sist et al., 1998; Pereira et al., 2002; Peña-Claros et al., 2008; Rockwell et al., 2007), the effects of logging at the community and/or smallholder scale has been little studied. Furthermore, research about the effects of logging on natural regeneration for multiple-use forest management (i.e., non timber forest products-timber) is minimal. More specifically, no published studies have looked at the effects of current logging practices—formal and informal—on natural

regeneration and extent of disturbance in community forests rich in Brazil nut (*Bertholletia excelsa*).

I investigated natural regeneration (individuals  $\leq 10$  cm dbh) responses of 10 logged timber species, and 2 economically important non-timber species (Brazil nut; and rubber, *Hevea brasiliensis*) to unlogged forest, formal, and informal logging. I characterized regeneration densities of examined species by type of disturbance (i.e., skid trails, access roads, felling gaps, and log landings), and location of individual plants within disturbed areas (edge and core zone). Additionally, the extent of disturbance caused by formal and informal logging was estimated to compare differences in extent of logged area, number of logged species, logging intensity and percent of disturbed area between logging types (formal and informal).

This thesis has a main chapter (Chapter 2) and two additional chapters (Chapters 1 and 3). While Chapter 2 is a single, independent chapter prepared for submission to a peer-reviewed journal, Chapters 1 and 3 provide a general contextual introduction and conclusions of the main themes of this paper, respectively.

CHAPTER 2  
THE GROWING DILEMMA OF TIMBER HARVESTING IN BRAZIL NUT RICH,  
COMMUNITY FORESTS IN NORTHERN BOLIVIA: EFFECTS ON NATURAL  
REGENERATION AND FOREST DISTURBANCE

**Introduction**

Interest in multiple-use forest management (MFM) that integrates management for diverse products such as timber, non-timber forest products (NTFPs), and environmental services has increased globally in recent years (García-Fernández et al., 2008). This holds particularly true in tropical regions where high biodiversity and carbon sequestration intersect with rural communities dependent on these forest resources and the ecosystem services they provide (Smith and Scherr, 2003).

MFM has been practiced for centuries by rural Amazonian inhabitants with the non-timber sector historically driving regional economies. In recent years, Brazil nut (*Bertholletia excelsa*: Lecythidaceae family) has emerged as a key economic species for rural communities, particularly in Northern Bolivia where it not only provides cash and subsistence income, but also temporary nut collection and factory-based processing jobs during the harvest season (Guariguata et al., 2008). As of 2006, Bolivia exported products of 66 forest species: 61 timber, and 5 non-timber species. Brazil nut was the single-most economically important forest export species, constituting 39% (US\$ 73 of \$ 187 million) of the forest-based national economy (Cámara Forestal de Bolivia, 2007). Furthermore, both Brazil nut and timber exports are steadily rising (Cámara Forestal de Bolivia, 2007), parallel to increased logging pressure on these Brazil nut rich forests (Cronkleton et al., 2009), often held on communally-titled lands. The department of Pando serves as a nexus for these Brazil nut-timber interactions where 245 peasant and 5 indigenous communities have legal title to 2 and 1.4 million ha of forest, respectively (Pacheco et al., 2009).

Prior to Bolivia's 1996 forestry law (Ley 1700), forest dependent communities had no legal rights over the timber growing on the forests they inhabited. Timber rights were solely granted to large-scale timber enterprises (Pacheco, 1998). The new legislation recognized the legitimacy of traditional communities, as well as a broad range of forest stakeholders, to timber harvest rights. These rights came with the legal responsibilities of developing a government-approved General Forest Management Plan (GFMP) and Annual Operational Logging Plans (AOLPs). Within this legal framework, a 20–25-year harvest rotation is required, except in forest areas < 200 ha, in which case, a 5-year minimum harvest rotation is recommended, but not required. They must also follow a series of logging restrictions and forest management guidelines known as Reduced Impact Logging (RIL), developed to enhance the likelihood that timber yields and forest integrity are sustained over the long term (Putz et al., 2000; Fredericksen and Putz, 2003; Zarin et al., 2007). AOLPs must be submitted each harvest year to obtain permits for timber extraction from a particular compartment area corresponding to a specific year stated in the GFMP. Logging is considered “legal” upon formal approval of these plans. Few communities, however, possess the capacity and capital to comply with the requirements of the forestry law by themselves (Martínez Montaña, 2008). Thus, instead of solely conducting formal logging, many communities also engage in informal logging, commonly described as illegal logging.

In contrast to the planning and harvest restrictions that characterize formal (legally-approved) logging, informal logging is mainly based on extracting a few high value tree species (Table 2-1), such as mahogany (*Swietenia macrophylla*), cedar (*Cedrela odorata*), amburana (*Amburana cearensis*), and marfil (*Aspidosperma macrocarpon*). In this system, landholders dispense with a formal inventory and road planning, opting instead to open short paths to target

trees they intend to sell to loggers (Table 2-1). These paths are often then used by loggers to extract the timber, either in the form of *cuartones* (trunk sliced into sections), or as logs disguised with legal documentation from a separately approved AOLP submitted with overestimated volumes (Superintendencia Forestal, 2007; Martínez Montaña, 2008). At the time of our fieldwork in 2009, though not comfortable with, nor perhaps even aware of breaking the law, most community-based landholders were engaged in informal logging activities. Their reasoning was that over the long-term, formal logging could not guarantee a steady provision of economic benefits. They also noted that revenues from informal logging were immediately available in difficult times such as illness or needed land or house improvements, whereas benefits from formal logging were often delayed.

To what extent are these growing logging activities affecting the Brazil nut populations that serve as the economic mainstay of Pando's forest-based communities? In this study, we address a set of questions to examine the effects of logging on new recruits and survivors post logging  $\leq 10$  cm dbh (hereafter referred as to regeneration) densities of Brazil nut (*B. excelsa*), rubber (*Hevea brasiliensis*), and 10 timber species in two formally- and four informally-logged sites. Based on previous studies of secondary forests (Cotta et al., 2008) and gaps (Myers et al., 2000; Zuidema and Boot, 2002) that highlight a favorable response by *B. excelsa* to disturbance, we hypothesized that formal and informal logging would enhance *B. excelsa* regeneration densities compared to those encountered in unlogged forest. Different types of logging disturbances (i.e., skid trails, access roads, felling gaps, and log landings) in both formal and informal logging sites were also differentiated. To further explain mechanisms behind *B. excelsa* regeneration, we quantified the following variables within each disturbance type: core and edge zones of disturbed areas (hereafter referred as microsite zone), distance to nearest conspecific

potential mother tree, and canopy openness. Similar questions were also addressed for *H. brasiliensis*, and 10 commercially important timber species. To better understand differences between formal and informal logging, we compared the extent of logged area, number of logged species, logging intensity, and percent of disturbed area. Some studies have demonstrated that RIL, such as required under formal logging in Bolivia, can reduce residual damage up to 50% compared to conventional informal (or illegal) logging (Pinard and Putz, 1996; Johns, 1996; Sist et al., 1998). Thus, we hypothesized that informal logging will cause greater disturbance than formal logging.

### **Study Area**

This study was implemented in six community landholdings in the province of Filadelfia, western of the department of Pando, Bolivia (Figure 2-1). Pando receives average annual rainfall of 1700 to 2000 mm, has a mean temperature of 26°C, and a three-month dry season (Mostacedo et al., 2006). Soils are classified as ferrasols (oxisols), with low fertility and high aluminum concentration. The diversity of tree species  $\geq 10$  cm dbh ranges between 52 and 122 species ha<sup>-1</sup>, and average tree abundance ranges between 544 and 627 trees ha<sup>-1</sup> (Mostacedo et al., 2006). Deforestation in Pando remains low compared to neighboring states in Brazil and Peru (Pacheco, 1998; Duchelle, 2009), attributed to its almost complete isolation from the rest of the country and to its long history of NTFP extractivism. Of the entire Bolivian Amazon, 50% is under collective ownership, and over 257 communities have been granted ~500 ha per family of Brazil nut rich forests (Guariguata et al. 2008).

The province of Filadelfia, Pando (our study site), is a hot spot for informal logging. A good proportion of informally extracted logs are confounded with legally extracted timber to fulfill the authorized timber volumes from a managed area, and are then destined to carpentries installed in nearby cities or used in local building construction (Pacheco et al., 2009).

## Methods

### Field Data Collection

Fieldwork was conducted in six forest sites logged between 2004 and 2007, 2 to 5 years prior to data collection; two sites were formally logged and four were informally logged. Each logged site was spatially independent from the others, except for one site on which both formal and informal logging overlapped, but effects of each logging type could be evaluated independently.

Within each logged site, all types of logging disturbances were easily distinguished and mapped using a Venture HC Garmin GPS; length of access roads and skid trails were calculated using Garmin MapSource software. Formal logging sites had a pre-harvest defined area as a prerequisite to meet forest management criteria in Bolivia's 1996 forestry law. In contrast, informal logging areas were not defined as in a formal management plan. To delineate these ill-defined areas we used terrain observations and satellite imagery maps (i.e., Google EARTH PRO) to first identify the extent and location of logging induced disturbances such as skid trails, access roads, felling gaps, and log landings. Second, using both maps and ground inspection, we identified natural features such as wetlands, rivers, and/or slopes that often define borders of logged sites. We then integrated logging-induced disturbance information with that provided by natural features to define approximate borders for informally logged sites. To test the veracity of this border delineation method, we followed the same procedure on our formally logged areas, and found that the resulting area using this method differed from the original, predetermined area listed on the formal management plan by less than 1%. This confirmed that our border delineation method was realistic and trustworthy for application in determining extent of disturbance on a per hectare basis.

To compare differences in regeneration densities between unlogged, formally-logged, and informally-logged sites, we established 7 to 10 plots in each disturbance type (skid trails, access roads, felling gaps, and log landings) within each logged site; and 9 to 10 plots within unlogged areas at each logged site (Figure 2-2). In total, we established 203 plots allocated as follows in each disturbance type (and formal and informal logging sites): 40 on skid trails (10 formal, 30 informal), 39 on access roads (19 formal, 20 informal), 57 in felling gaps (21 formal and 36 informal), 10 in log landings (6 formal, 4 informal), and 57 plots in unlogged areas (20 formal, 37 informal). The 7 to 10 felling gaps were randomly selected after identifying all felling gaps in each site; plot size in these gaps varied by extent of the disturbance caused by the felling tree. Plots in skid trails and access roads were systematically established every 25 m along the course of these disturbance types; plots were 25 m long, and plot width was determined by the disturbance itself. Plots were installed in all log landings, with plot size varying according to the extent of the landing. The 9 to 10 plots in unlogged forests were located at a random direction typically from each sampled felling gap. Unlogged plots were 25 x 25 m, and were always located at least 50 m from any logging disturbance.

To measure the extent of each disturbed plot, we extended a metric tape through the middle of the longest axis of each disturbance type, marking every 5 m along that axis (Figure 2-2). Then, we measured the perpendicular distance from each of these marked points, placing a flag at the edge of the disturbance limit (Contreras et al., 2001). The total area of each disturbance type included not only the core disturbed area but also an edge zone that extended 2 (skid trails and access roads) to 3 m (felling gaps and log landings) into the forest (Figure 2-2). The core zone was defined by the edge of open overhead canopy, and the edge zone was defined as the area beyond that, where some indirect sunlight penetrated into the forest edge due to logging

disturbance. Edge zones were also distinguishable from undisturbed forest by the presence of coarse woody debris either caused by the felling of the logged tree or by the maneuvering of logging machinery.

To calculate the total area (core + edge zone) of each felling gap and log landing, we followed the method used by Contreras et al. (2001), dividing the whole area into two sub-areas at each side of the middle axis:

$$A1 = d * [((d_{left\ 1} + d_{left\ n}) / 2) + d_{left\ 2} + d_{left\ 3} + \dots + d_{left\ n-1}] \quad \text{(Equation 2-1)}$$

$$A2 = d * [((d_{right\ 1} + d_{right\ n}) / 2) + d_{right\ 2} + d_{right\ 3} + \dots + d_{right\ n-1}] \quad \text{(Equation 2-2)}$$

Where:

$d$  = distance interval along the largest middle axis between two perpendicular distances

$d_{left\ 1} + \dots + d_{left\ n}$  = distances from the middle axis to the left side of the plot

$d_{right\ 1} + \dots + d_{right\ n}$  = distances from the middle axis to the right side of the plot

Subsequently, we measured canopy cover of the core disturbed area with a concave spherical densiometer held at 1m height (Lemon, 1957). The exact location of the measurements varied according to disturbance type. For example, in tree fall gaps, measurements were taken at the stump, trunk, and crown zone of the felled tree, and were averaged to obtain mean percent canopy openness by plot. On skid trails and access roads, measurements were taken at 5 and 20 m along the 25 m length of the middle axis. On log landings, we took one measurement in the center of the middle axis. The formula used to calculate percentage of canopy cover was as follows:

$$\text{Canopy Cover \%} = 100\% - (\# \text{ of empty squares} \times 4.17) \quad \text{(Equation 2-3)}$$

We obtained percent canopy openness by subtracting the percent canopy cover from 100%.

In each plot, we carefully searched and counted all regeneration (new recruits and survivors post logging  $\leq 10$  cm dbh) of 2 non-timber, and 10 timber species. We recorded height and stem diameter at the base if plant height was  $\leq 1.5$  m; otherwise, we measured dbh. We also recorded microsite zone (core and edge) within the plot of each measured plant in each

disturbance type. However, when the number of individuals of any given species was  $> 50$ , we measured diameter, height, and microsite on only the first 50.

To assess whether distance to potential mother tree affects regeneration densities, potential mother trees were searched up to 50 m starting from the border of the plot. This distance was originally selected for *B. excelsa* based on Cotta et al.'s (2008) method regarding maximum seed dispersal, and was standardized and used for all species. For *B. excelsa*, potential mother trees were considered to be  $\geq 40$  cm dbh (K.A. Kainer, personal communication). Distance to, and dbh of the nearest potential mother tree were recorded in all 203 plots regardless of whether regeneration of this species was encountered in the plot or not. Distance to, and dbh of, the nearest potential nearest mother trees of all other species were only recorded when individuals of that given species were found in the plot. For these species, potential mother trees were considered to be those  $\geq 20$  cm dbh, applying a standardized diameter that approximates when most Amazonian tree species start producing fruits (Van Rheenen, 2005). In these cases, the influence of mother tree then responded to the question: Given the presence of regeneration of a given species, are densities explained by distance to the nearest potential mother tree?

### **Data Analysis**

All statistical analyses were performed using SAS software (Version 9.2), and differences were considered statistically significant at  $p$ -values  $\leq 0.05$ . Because the sample size to determine differences on extent of disturbance between logging treatments (2 formal and 4 informal) were small, the two-sample non parametric test (PROC NPAR1WAY WILCOXON) was performed to test differences between logged areas, number of logged species, logging intensities (trees  $\text{ha}^{-1}$  and  $\text{m}^3 \text{ha}^{-1}$ ), and percent disturbed area. For this, the continuity correction factor assumed in the computation of the standardized  $Z$  test statistics was removed using the “correct = no” statement in the SAS procedure.

A Generalized Linear Mixed Modeling procedure PROC GLIMMIX was used to predict regeneration densities by species as a function of logging type, distance to nearest potential mother tree, disturbance type, and microsite. However, due to low abundance of some species, the usage of this model was restricted to only 9 of the 12 examined species, 2 non-timber and 7 timber species. The response variable (number of recruits and survivors post logging = regeneration density) was characterized as a Poisson distribution, using natural logarithm of plot area as an offset to properly reflect differences in plot sizes. To avoid statistical issues associated with overdispersion due to masting events of some species, regeneration density per sample unit included in the analysis was cut off at 1000 individuals, affecting only two species: *H. brasiliensis* and *Astronium lecontei*, each masting event occurring in a different sample unit. Least squares means were also generated, and adjusted using Scheffe's method to account for multiple comparisons between logging types, disturbance types, microsite zones, and distance classes to nearest potential mother tree.

Three models were explored to explain *B. excelsa* regeneration density. Model 1 tested whether the three logging treatments (unlogged, formal, and informal logging), canopy openness, and/or distance to nearest potential mother tree explained regeneration densities (N = 196). For purposes of analysis, distances to nearest potential mother trees were classified in seven classes: A first class was set for trees encountered within the plot, five represented sequential 10 m intervals, and a sixth for distances > 50 m from the border of each plot. Model 2 explored the effects of type of logging disturbance (felling gap, skid trail, access road, and log landing), and therefore excluded plots in unlogged sites (thus, N = 143) from Model 1. Finally, Model 3 only used those plots where *B. excelsa* individuals were present (N = 49), to explore whether microsite zone helped explain densities (Table 2-3).

Similar methods and logic used for *B. excelsa* were followed to model regeneration densities of all other species. Because distance to nearest potential mother tree for all other species was only recorded when individuals of that particular species was encountered in a particular plot, a most basic model (Model 0) was run. Thus, Model 0 only accounted for two explanatory variables, logging treatments and canopy openness, to obtain differences between logging treatments based on a larger sample size (195–196 plots). Models 1 and 3, which used potential mother tree as an explanatory variable, considered the logged species as a potential mother tree for individuals encountered within examined felling gaps. For some of these species (*H. brasiliensis*, *C. odorata*, *Hymenaea parvilofia*, and *Mezilaurus itauba*), Model 3 showed slightly high chi-square statistics, indicating poor model fit. However, alternative models to Model 3 did not improve the chi-square statistic, and although residuals indicated no heteroscedasticity, results should be viewed conservatively. The effects of formal and informal logging on regeneration density of three conspecific logged species, as well as the proportion that each species represented by the twelve examined species found in the 203 plots, were compared using descriptive statistics in Microsoft ® Office Excel 2007.

## Results

### Disturbance Responses to Formal and Informal Logging

Although logging intensity did not differ between formal and informal logging (0.28 vs 0.24 trees ha<sup>-1</sup>,  $p = 0.184$ ; 1.8 vs 1.6 m<sup>3</sup> ha<sup>-1</sup>,  $p = 0.486$ ), formal logging resulted in more species logged in a given logging site ( $61 \pm 3.0$  SE vs.  $23 \pm 8.4$  SE;  $p = 0.047$ ) and a larger portion of disturbed area ( $10.6\% \pm 0.6$  SE vs.  $6.9\% \pm 1.26$  SE;  $p = 0.047$ ) than informal logging (Table 2-2). Despite no differences between formal and informal logging intensities, formal logging sites had the greatest percent of disturbed area (Figure 2-3).

## Explaining Brazil nut Regeneration

Relative regeneration abundance (individuals  $\leq 10$  cm dbh) of *B. excelsa* was very low proportional to a pool of twelve species (0.67% of 100%) (Figure 2-4). Regeneration densities did not differ between logging types ( $3.9 \pm 1.3$  ind. ha<sup>-1</sup> in unlogged,  $7.6 \pm 1.7$  ind. ha<sup>-1</sup> in formally-, and  $5.8 \pm 1.2$  ind. ha<sup>-1</sup> in informally-logged sites, respectively;  $p = 0.315$ ) (Figure 2-5; Table 2-3) nor with distance to potential mother trees ( $p = 0.477$ ) (Table 2-3). Differences were, however, observed between disturbance types ( $p = 0.0272$ ) (Table 2-3). Specifically, regeneration in log landings ( $15.1 \pm 7.2$  ind. ha<sup>-1</sup>) was significantly higher than in skid trails ( $2.9 \pm 1.2$  ind. ha<sup>-1</sup>) (Figure 2-6). Interestingly, differences between microsites (core versus edge of disturbed area) were observed for all species other than *B. excelsa* ( $14.6 \pm 3.0$  vs  $17.1 \pm 3.4$  ind. ha<sup>-1</sup>;  $p = 0.444$ ) (Figure 2-7; Table 2-3).

## Explaining Regeneration of Other Economically Important Species

Regeneration of *Astronium lecontei*, *C. odorata*, *H. brasiliensis*, and *M. itauba* were four of the most abundant species found within all plots (unlogged, formally- and informally-logged) and across all types of disturbances (Figure 2-4). In contrast, individuals of *A. cearensis* and *S. macrophylla* were rare, representing only 0.04% and 0.21% of the 12 studied species respectively, and these species were only found in felling gaps where conspecific trees were felled (Figure 2-4).

When comparing regeneration densities between logging treatments, only densities of three species varied between unlogged and formally-logged sites; *C. odorata* ( $8.6 \pm 2.9$  vs  $39.8 \pm 9.3$  ind. ha<sup>-1</sup>;  $p = 0.001$ ) and *Dipteryx odorata* ( $3.0 \pm 1.4$  vs  $14.1 \pm 4.4$  ind. ha<sup>-1</sup>;  $p = 0.030$ ) had higher densities in formally logged sites, whereas *M. itauba* had higher densities in unlogged sites ( $114.0 \pm 18.5$  vs  $60.4 \pm 9.5$  ind. ha<sup>-1</sup>;  $p = 0.017$ ) (Figure 2-5; Table 2-4). In all three cases, densities did not differ between formally- or informally-logged sites (Figure 2-5).

Some species demonstrated differences in regeneration densities between types of logging disturbances. *Aspidosperma vargasii* presented lower densities on skid trails ( $31.1 \pm 8.4 \text{ ind. ha}^{-1}$ ) and access roads ( $15.7 \pm 4.3 \text{ ind. ha}^{-1}$ ) than in felling gaps ( $84.2 \pm 16.6 \text{ ind. ha}^{-1}$ ) ( $p = <.0001$ ). *D. odorata* had higher densities in log landings ( $198.1 \pm 136.7 \text{ ind. ha}^{-1}$ ) than on skid trails ( $7.0 \pm 3.1 \text{ ind. ha}^{-1}$ ), access roads ( $10.4 \pm 3.9 \text{ ind. ha}^{-1}$ ), and felling gaps ( $17.1 \pm 5.1 \text{ ind. ha}^{-1}$ ) ( $p = 0.002$ ). Higher regeneration densities of *Hymenaea parvifolia* were observed in felling gaps ( $24.2 \pm 6.7 \text{ ind. ha}^{-1}$ ) and log landings ( $103.7 \pm 70.7 \text{ ind. ha}^{-1}$ ) than on skid trails ( $2.7 \pm 1.2 \text{ ind. ha}^{-1}$ ) and access roads ( $2.9 \pm 1.3 \text{ ind. ha}^{-1}$ ) ( $p = <.0001$ ). Similarly, *M. itauba* also had higher densities in felling gaps ( $173.2 \pm 23.2 \text{ ind. ha}^{-1}$ ) and log landings ( $227.8 \pm 78.1 \text{ ha}^{-1}$ ) than on skid trails ( $36.2 \pm 6.9 \text{ ind. ha}^{-1}$ ) and access roads ( $40.2 \pm 7.2 \text{ ind. ha}^{-1}$ ) ( $p = <.0001$ ) (Figure 2-6; Table 2-4).

Regarding regeneration microsite preferences, all species had higher densities at the core than the edge of disturbed areas ( $p \leq 0.05$ ) (Figure 2-7; Table 2-4). Recruitment densities of *A. vargasii* ( $p = 0.024$ ), *A. lecontei* ( $p = 0.001$ ), *C. odorata* ( $p = 0.003$ ), *D. odorata* ( $p = 0.014$ ), *H. parvifolia* ( $p < 0.0001$ ), and *T. impetiginosa* ( $p = 0.001$ ) differed with distance to nearest potential mother tree (Table 2-4). Greater densities were observed at shorter distances, and then gradually decreased at further distances (Figure 2-8).

Individuals from the three most dominant species were identified within felling gaps, to specifically determine whether the species logged affected regeneration of a particular species. Regeneration of two species, *M. itauba* and *A. vargasii*, were always among the three most abundant species (reaching up to 70 and 47% of all species), regardless of the species logged. Regeneration from *C. odorata*, *D. odorata*, and *H. parvifolia* were consistently among the three most abundant in conspecific felling gaps where these species were logged, 41%, 34% and 43%,

respectively, indicating a high degree of conspecificity in regeneration establishment. Only three species were logged in both formal and informal logging sites (*A. cearensis*, *C. odorata*, and *D. odorata*). Of these, only *C. odorata* showed differences—a higher proportion of regeneration—in formally logged than in informally logged sites (Figure 2-9).

## Discussion

### The Brazil nut Case

Despite being considered a gap dependent species (Myers et al. 2000; Zuidema and Boot 2002), and demonstrating comparatively higher regeneration densities (individuals  $\leq 10$  cm dbh) in abandoned agricultural fallows than in mature forests (Cotta et al., 2008), we found no differences in *B. excelsa* regeneration densities between unlogged and logged (neither formal nor informal) sites. However, higher densities were observed in larger logging disturbances (i.e., log landings vs skid trails) (Figure 2-6). Still, no differences were found in *B. excelsa* preferences regarding microsite zone; the higher incidence of light in the core of disturbed plots did not translate into higher regeneration densities than within edge zones. This finding may be a reflection of the habitat preferences of seed dispersers (Chen et al. 1999). The logging debris usually encountered on the edges of logging-induced disturbances provides a measure of safety to seed dispersers such as agouti (*Dasyprocta spp.*) while they eat and bury *B. excelsa* seeds. Individuals of this species are often found within accumulated debris or behind a tree stem (Peres and Baider, 1997; personal observation, M. Soriano). The lack of relationship between regeneration density and distance to nearest potential mother trees contrasts with findings of Cotta et al. (2008), who found a positive correlation between proximity to mother trees and *B. excelsa* regeneration densities. Lack of effects could be explained by the high density of adult trees in our sites, and/or that seed reburial at long distances by agoutis played an important role in seed dispersal (Tuck Haugaasen et al., 2010) and thus on its establishment. Our overall finding

that low logging intensity did not negatively affect regeneration of this keystone species complements Guariguata et al.'s (2009) results of minimal logging damage to *B. excelsa* trees  $\geq$  10 cm dbh in certified forest concessions in which RIL guidelines were followed.

### **The Case of Timber Species**

Logging under the conditions of our study sites resulted in greater regeneration of two important species; both, *D. odorata* and *C. odorata* had higher densities in logged areas than in unlogged forest. In addition, regeneration densities of all timber species tended to be higher on log landings and in felling gaps than on skid trails and access roads. This finding is consistent with others that demonstrated that many commercial timber species respond favorable to higher light environments (Licona Vasquez et al., 2007; Peña-Claros et al., 2008). We also found that, with the exception of *A. lecontei*, canopy openness positively affected regeneration densities of all timber species examined, as well as *H. brasiliensis*. These species also had higher densities in the higher light microsites (core vs edge of disturbed areas).

Some species (e.g., *A. cearensis*, *C. odorata*, and *D. odorata*) showed high regeneration densities of conspecific logged species, probably because seedfall of these species coincided with logging season in Northern Bolivia. This suggests that the time of year when species are logged (mid-to-end of the dry season) can be planned to stimulate regeneration establishment (Mostacedo and Pinard, 2001; Soriano Candia, 2005). The fact that individuals of two of the most important timber species, *A. cearensis* and *S. macrophylla*, were rarely encountered in the study plots (each representing less than 1% of total counted individuals from all species) is attributed to historical logging intensity of these high value species (Putz et al., 2001) that has depleted them over time. Regeneration of both species only occurred in felling gaps where these species were logged, implying that population recovery may only occur once selective logging of

these species is halted followed by intervention with silvicultural treatments (Fredericksen and Pariona, 2002).

### **Disturbance Created by Formal and Informal Logging**

Our hypothesis was not supported on that the extent of logging disturbance would be greater in informally (illegally) logged sites than in those with approved management plans (formally logged) as documented in multiple studies (Johns, 1996; Pinard and Putz, 1996; Sist et al., 1998; Pereira et al., 2002). Contrary to expectations, we found that formally logged sites incurred a greater percentage of disturbed area than informally logged sites. While previous studies were carried out within large (industrial) scale logging operations, logging activities in our communally owned sites were relatively small-scale. The observed comparatively lower extent of disturbance in our informally logged sites may be attributed to two reasons; first, the opening of short paths to reach target trees tended to reduce felling of non-target trees. A study of small-scale RIL logging within communities in Acre, Brazil also attributed their finding of no effects from marking future crop trees to the fact that landholders marked paths to guide skidding activities (Rockwell et al., 2007). Secondly, logging intensities, and number of trees logged, were lower in informal than in formal logging areas.

### **In the Shoes of Community-Based Landholders**

Undoubtedly, formal logging has the potential to return significant economic benefits to forest-dependent communities (Benneker, 2006); however, under the circumstances of the community landholdings we examined, this was not entirely evident. First, formal logging resulted in a greater relative disturbed area than informal logging, and a greater number of trees logged, perhaps more than was necessary to maintain a family's livelihood at a given year. Indeed, formal logging (versus informal logging) often results in extraction of greater timber volumes; according to the current forestry law, up to 80% of all trees above the minimum

diameter cutting can be removed from a managed forest. Formal logging may also threaten the long-term economic sustainability of a family's forest. A family's entire productive forest is usually around 200 ha; therefore, formal timber harvesting is often carried out in a single year, extracting most commercial stems, to cover the greater planning requirements and related costs dictated by formal logging. (Cronkleton et al., 2009). A lack of sufficient areas to harvest in subsequent years could increase pressure over the originally-logged area to harvest seed trees (C. Borobobo, personal communication), although new species, for which there was no market demand during the first logging, may also be tapped in subsequent years. Finally, in practice, not all felled trees that are formally logged in any given year make it to the market. This translates into substantial economic losses for smallholders, because third parties do not pay for those logs that remain in landholders' forest to rot (Benneker, 2006; P. Paz and C. Borobobo, personal communication). Thus, Martínez Montaña (2008) reports that community people conclude that formal logging can be worse than informally selling smaller quantities of high-value, illegally logged timber species.

Indeed, formal logging may also affect smallholder livelihoods negatively by: (1) indirectly prohibiting small-scale timber production; (2) requiring complex paperwork that forest people are uneasy to complete by themselves, (3) prohibiting traditional activities such as hunting, although we certainly acknowledge that not all game harvests are sustainable in managed forests for timber, and, (4) advocating that creation of no-take zones within communities is the best alternative for forest conservation. From the community standpoint, it has become nearly impossible to comply with GFMP requirements without technical assistance and economic support of external institutions (Benneker, 2006; Martínez Montaña, 2008; Cronkleton et al., 2009).

Furthermore, roads built under formal logging regimes are often used as a conduit for informal loggers and wildlife poachers to access remote areas that were otherwise inaccessible. In reality, some formally managed forests are being degraded by informal loggers, leading indirectly to deforestation (Asner et al., 2006; personal observation, M. Soriano). In view of these constraints, informal logging may also be comparatively more compatible with community-based landholder's forest management and conservation objectives than formal logging as carried out under the current forestry regime.

### **Is Multiple-Use Forest Management Viable in Northern Bolivia's Communities?**

Findings that neither formal nor informal logging affected regeneration densities of *B. excelsa* within communally logged forests suggests that multiple-use, Brazil nut- timber forest management may be viable under certain circumstances. First, and perhaps most importantly, all our study sites were harvested at low intensities (0.13–0.31 trees ha<sup>-1</sup>), falling within the lowest range of logging intensities found in the literature (Jhons, 1996; Sist et al., 1998; Rockwell et al., 2007; Guariguata et al., 2009). Secondly, the extensive forest-based knowledge and the earlier described unforeseen management practices (i.e., guiding skid trails for informal loggers) carried out by community-based landholders could smooth the road towards sustainable management in practice. In our efforts to understand the dilemma on whether timber harvesting could be compatible with sustaining the keystone non-timber species *B. excelsa*, results suggest that both *B. excelsa* and most examined timber species regenerate successfully. Unexpectedly, neither formal nor informal logging proved totally incompatible with Brazil nut harvests. Nonetheless, despite low observed disturbance effects of informal logging (Rice et al., 1997), a sole focus on harvesting high value species which disregards its reproductive ecology, has been detrimental for the long-term population viability of those species (Figure 2-4).

We believe that governmental and nongovernmental regulations, policies and support mechanisms should promote small-scale rather than large-scale logging activities among community-based landholdings. Integrating traditional knowledge, and developing simplified and integral legal procedures and guidelines to improve the current forestry system (Martínez Montaña, 2008) would not only bring traditional forest users into the legal framework, but would also reduce deforestation and forest degradation and have positive implications for biodiversity conservation.

Table 2-1. Key variables that differentiate formal from informal logging in community-based landholdings in Northern Bolivia.

Issue	Formal logging	Informal logging
Timing of harvest	Every 20–25 years	Varies according to cash needs
Target species	Commercial species $\geq$ 50cm DBH	High value species only (1–5)
Remnant seed trees	20% of harvestable trees	Unknown
Inventory	Tree census	Exploratory
Basis for road planning	Maps, tree census	Seek shortest distance between trees
Capital needs	High	Low
Logging entity/loggers	Timber companies, sawmills	Experienced and inexperienced sawyers, landholders, sawmills, timber companies
Logged area*	> 200 ha	< 200 ha

\* Few exceptions may exist to given logged areas on the current table.

Table 2-2. Stand characteristics of formal and informal logging sites. Values with an asterisk indicate significant differences at one-sided  $p \leq 0.05$ , using the normal approximation for the two-sample Wilcoxon test.

Logging type	Logging site	Year of logging	Logging equipment	Logged area ha (SE)	#logged species (SE)	Logging intensity (SE)		% disturbed area (SE)
						trees ha <sup>-1</sup>	m <sup>3</sup> ha <sup>-1</sup>	
Formal	JS	2007	Skidder	225.6	7	0.28	1.6	11.2
	PP	2004	Caterpillar	203.8	10	0.28	1.9	9.9
<i>Mean</i>	<i>2</i>			<i>214.7</i> <i>(10.9)</i>	<i>8.5</i> <i>(1.5)</i>	<i>0.28</i> <i>(0.0)</i>	<i>1.8</i> <i>(0.15)</i>	<i>10.6</i> <i>(0.65)</i>
Informal	JS	2007	4-wheeled drive truck	271.0	4	0.13	1.3	5.3
	WT	2005	Skidder	153.6	5	0.25	1.2	7.2
	DG	2007	Skidder	25.6	1	0.27	1.4	6.1
	OC	2006	Skidder	32.0	3	0.31	2.5	9.1
<i>Mean</i>	<i>4</i>			<i>120.5</i> <i>(58.15)</i>	<i>3.3</i> <i>(0.85)</i>	<i>0.24</i> <i>(0.04)</i>	<i>1.6</i> <i>(0.30)</i>	<i>6.9</i> <i>(1.26)</i>
Pr(<0.05)	1-sided			0.1879	0.0468*	0.1845	0.4857	0.0468*
Pr(<0.05)	2-sided			0.3757	0.0937	0.3691	0.3757	0.0937

Table 2-3. Best models to explain density of individuals  $\leq 10$  cm dbh of *B. excelsa*. Different models were used to maximize the number of observations under the sampling scheme employed. Model 1 tested whether logging treatments (unlogged, formal and informal logging) and distance to nearest potential mother trees affected regeneration densities. Models 2 and 3 excluded unlogged plots to test whether densities differed between disturbance types (skid trails, access roads, felling gaps, and log landings) and microsite zones (core and edge of disturbances). Significant differences at  $p \leq 0.05$  are marked with an asterisk.

Model	# plots	Log-likelihood	Chi-square	Fixed effects				
				canopy openness	logging treatment	disturbance type	microsite	distance mother
1	196	762.69	0.69	0.0009*	0.3146			0.4768
2	143	544.88	0.71	0.0911	0.5536	0.0272*		
3	49	319.46	1.25	0.1430	0.8210	0.7832	0.4443	0.1672

Table 2-4. Best models to explain density of individuals  $\leq 10$  cm dbh of *H. brasiliensis* and seven timber species. Different models were used to maximize the number of observations under the sampling scheme employed. Models 0 and 1 tested whether regeneration densities differed between logging treatments (unlogged, formal and informal logging), and distance to nearest potential mother trees respectively. Models 2 and 3 excluded data from unlogged plots to test whether densities differed between disturbance types (skid trails, access roads, felling gaps, and log landings) and microsite zones (core and edge of disturbances). Significant differences at  $p \leq 0.05$  are marked with an asterisk.

Species	Model	# plots	Log-likelihood	Chi-square	Fixed effects				
					canopy openness	logging treatment	disturbance type	Microsite	distance mother
<i>Hevea brasiliensis</i>	0	195	982.60	0.61	0.4872	0.0945			
	1	36	119.99	1.00	0.1398	0.4072			0.0901
	2	142	703.07	0.61	0.4883	0.0386*	0.5769		
	3	29	379.21	5.89	0.4424	0.5714	0.5713	<.0001*	0.4100
Timber species									
<i>Aspidosperma vargasii</i>	0	196	724.65	0.99	0.0038*	0.4257			
	1	153	481.60	0.95	0.0926	0.9817			0.0238*
	2	143	525.89	0.95	0.2545	0.4685	<.0001*		
	3	109	874.89	2.02	0.0843	0.6277	0.362	<.0001*	0.0630
<i>Astronium lecontei</i>	0	195	790.85	0.89	0.4892	0.0448*			
	1	100	300.02	0.90	0.2990	0.0277*			0.0013*
	2	142	581.40	0.92	0.6629	0.2927	0.1930		
	3	80	599.28	1.69	0.8357	0.1360	0.8307	<.0001*	0.0070*
<i>Cedrela odorata</i>	0	195	791.56	0.87	<.0001*	0.0014*			
	1	114	368.50	0.93	<.0001*	0.1886			0.0033*
	2	142	560.42	0.94	<.0001*	0.1363	0.3227		
	3	102	1009.87	2.95	0.0083*	0.7126	0.6921	<.0001*	0.3043
<i>Dipteryx odorata</i>	0	196	888.19	0.74	<.0001*	0.0302*			
	1	66	202.05	0.91	0.0004*	0.0129*			0.0144*
	2	143	621.01	0.83	0.0141*	0.7204	0.0016*		
	3	59	469.59	1.87	0.0437*	0.0143*	0.0548	<.0001*	0.3641
<i>Hymenaea parvifolia</i>	0	196	858.95	0.80	0.9772	0.7283			
	1	80	238.44	0.80	0.2780	0.2582			<.0001*
	2	143	618.66	0.74	0.0020*	0.5707	<.0001*		
	3	57	604.26	2.80	0.2467	0.4041	0.3151	<.0001*	0.3672
<i>Mezilaurus itauba</i>	0	196	616.92	1.03	<.0001*	0.0169*			
	2	143	425.53	1.00	0.9570	0.0013*	<.0001*		
	3	125	924.71	2.19	0.0016*	0.0076*	<.0001*	<.0001*	0.7997
<i>Tabebuia impetiginosa</i>	0	196	906.66	0.49	0.0488*	0.0311*			
	1	34	82.18	0.76	0.4015	0.9708			0.0009*
	2	143	667.52	0.49	0.5008	0.0063*	0.0399*		
	3	28	218.55	1.74	0.6567	0.4835	0.0689	<.0001*	0.0006*

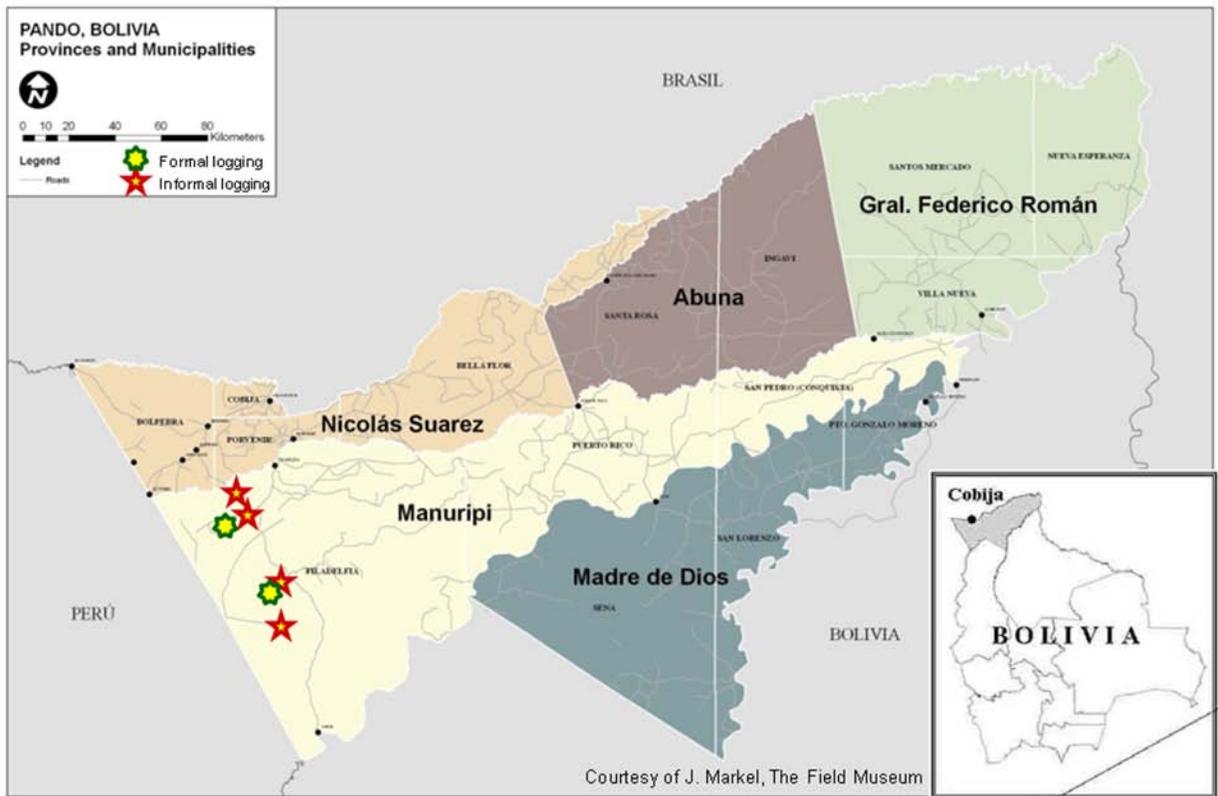


Figure 2-1. Geographic location of research sites in the department of Pando in the Bolivian Amazon.

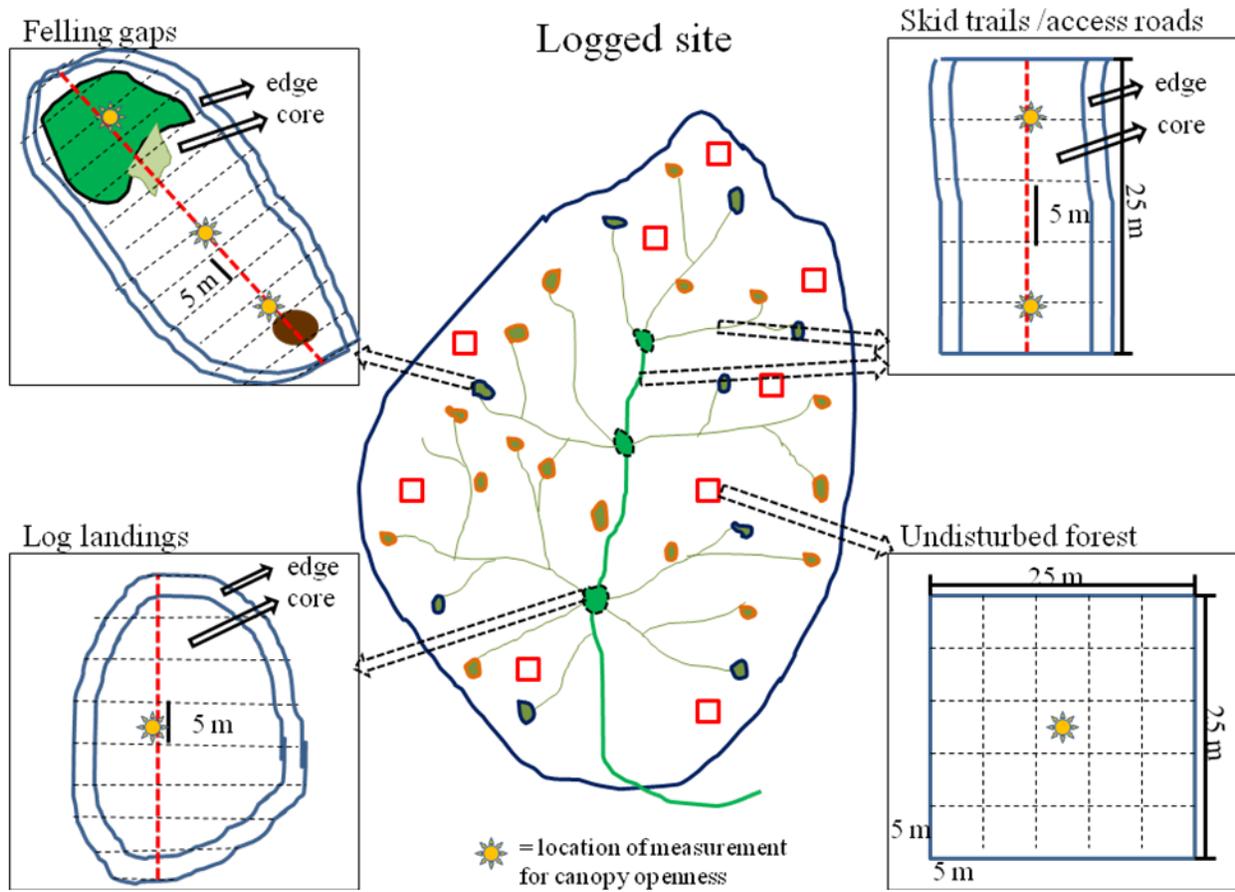


Figure 2-2. Sampling design within each logged site.

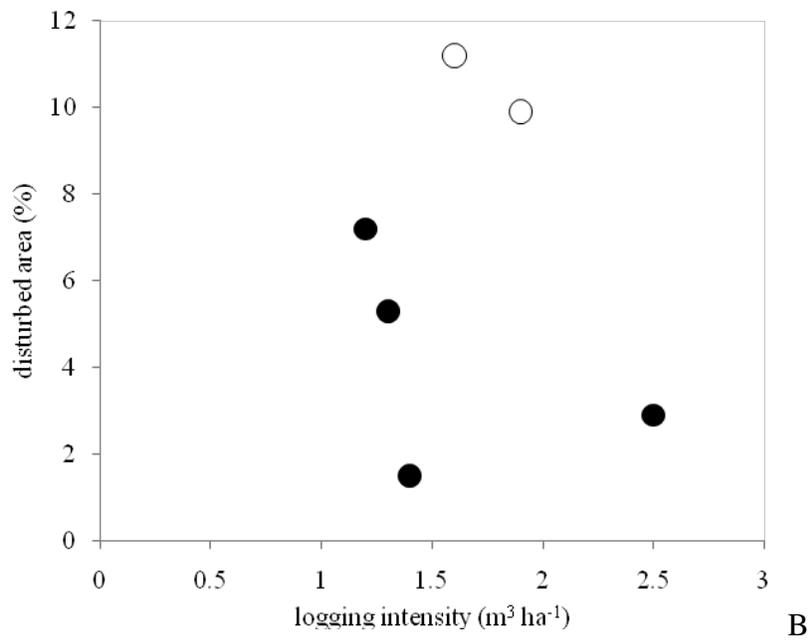
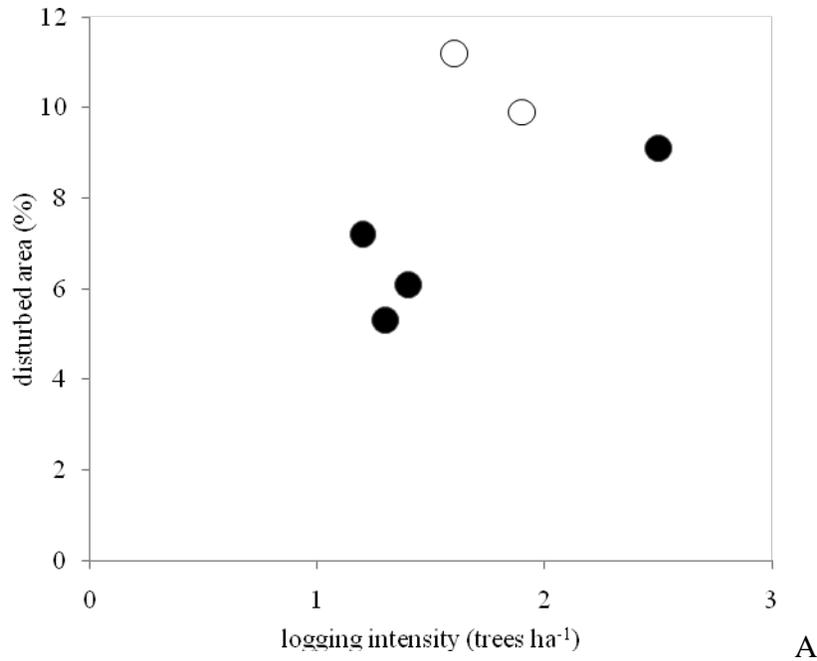


Figure 2-3. Percent disturbed area created by formal (filled circles) and informal logging (empty circles) as a function of logging intensity expressed as: A. trees ha<sup>-1</sup>, and B. m<sup>3</sup> ha<sup>-1</sup>.

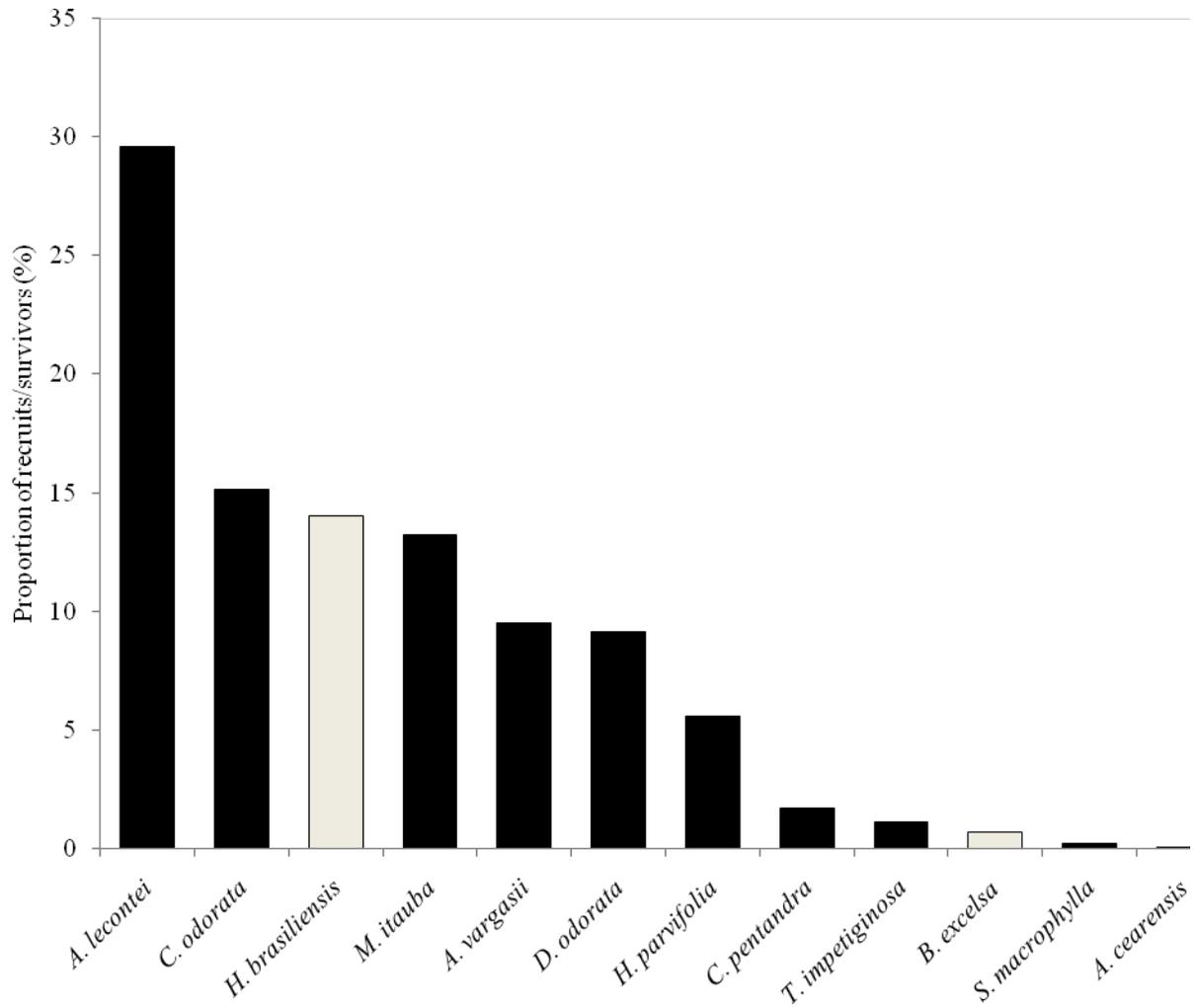


Figure 2-4. Proportion of individuals  $\leq 10$  cm dbh in six logged sites and all types of logging disturbance by species. Timber species are represented by filled bars, and non-timber forest products (NTFPs) species are represented by light-colored bars.

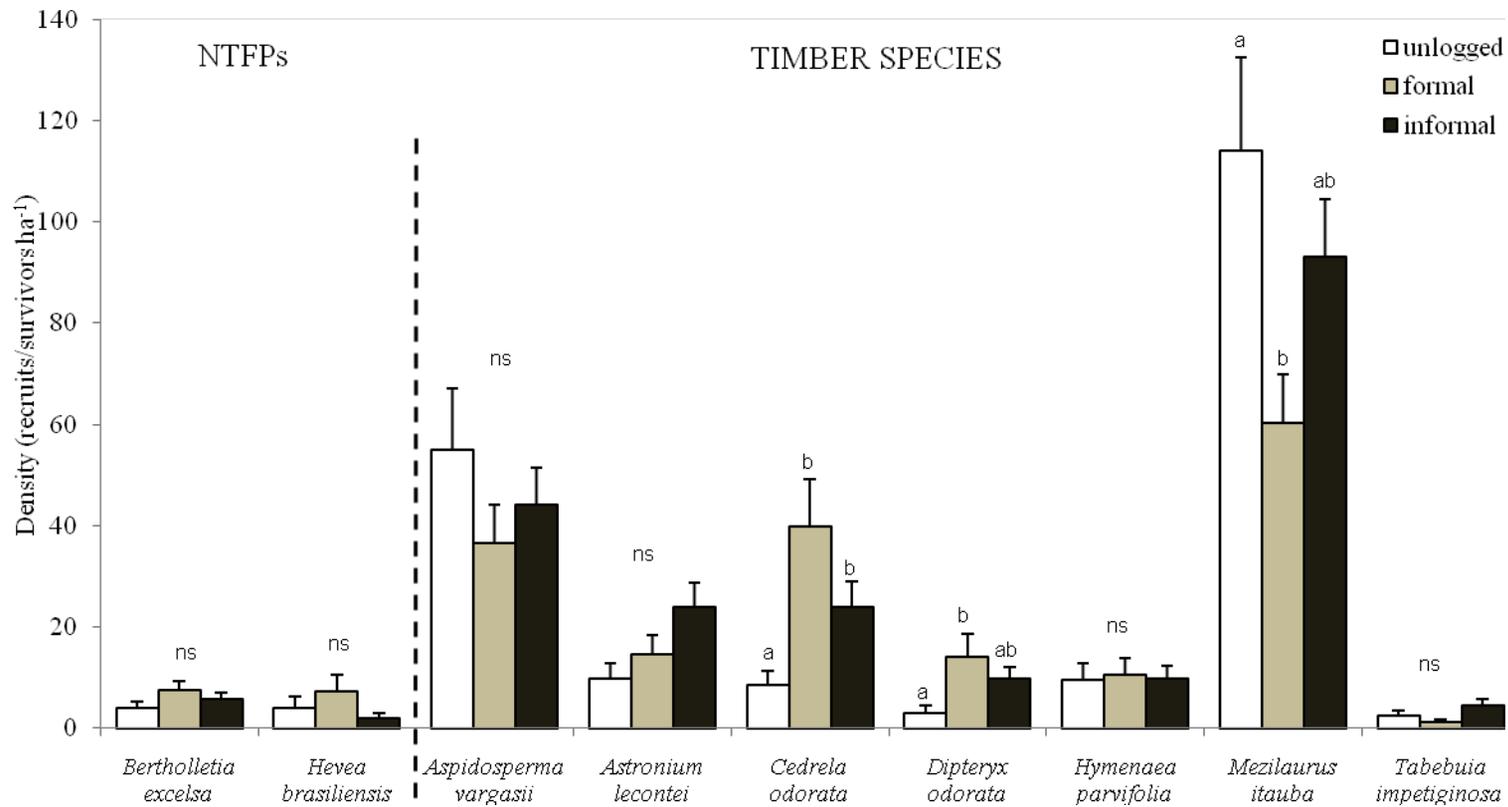


Figure 2-5. Density of individuals  $\leq 10$  cm dbh in unlogged forest, and formally and informally logged sites by species. Different letters represent statistical differences at  $p \leq 0.05$ ; ns = not significantly different.

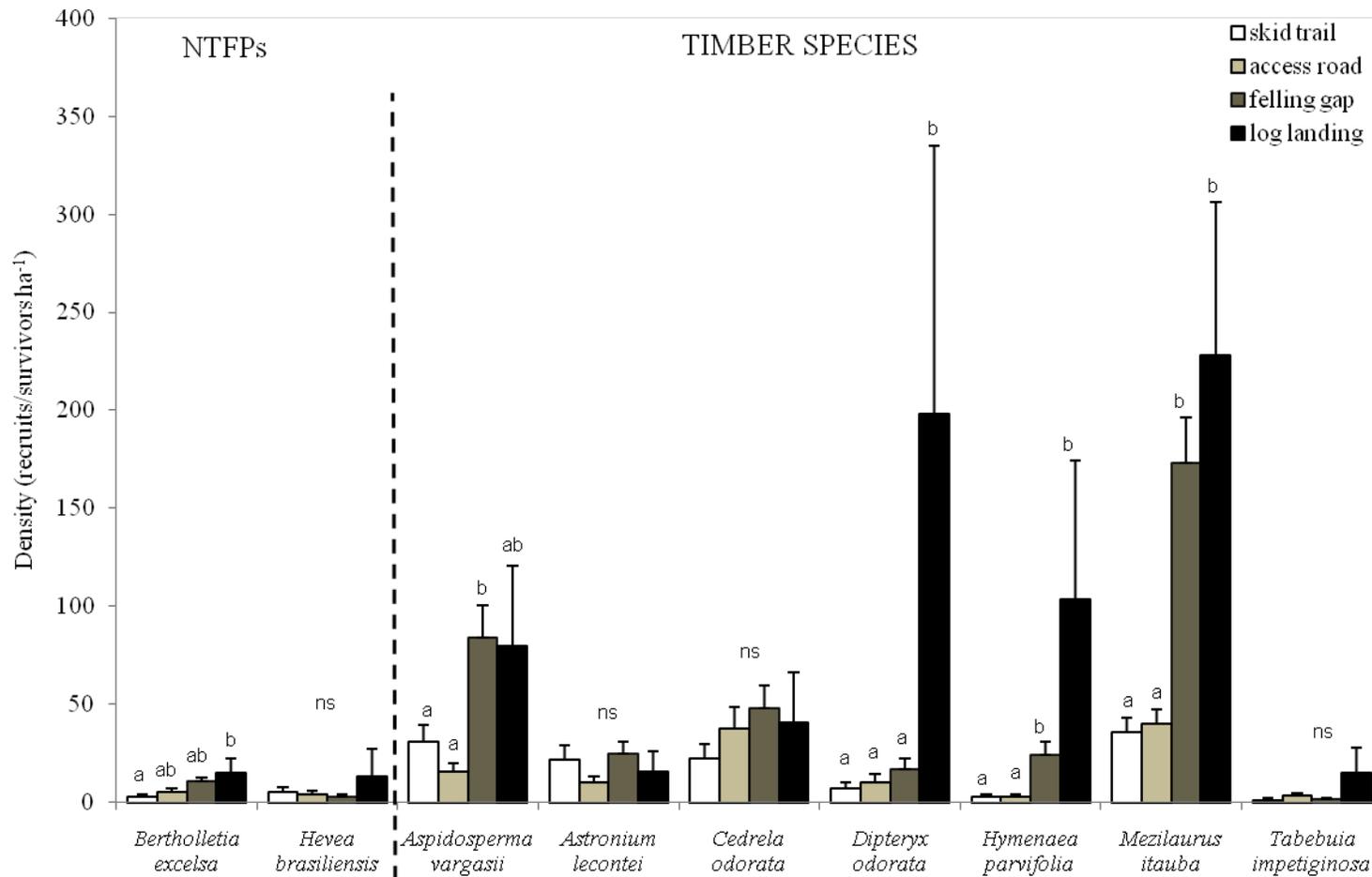


Figure 2-6. Density of individuals  $\leq 10$  cm dbh based on disturbance types and by species. Different letters represent statistical differences at  $p \leq 0.05$ ; ns, not significantly different.

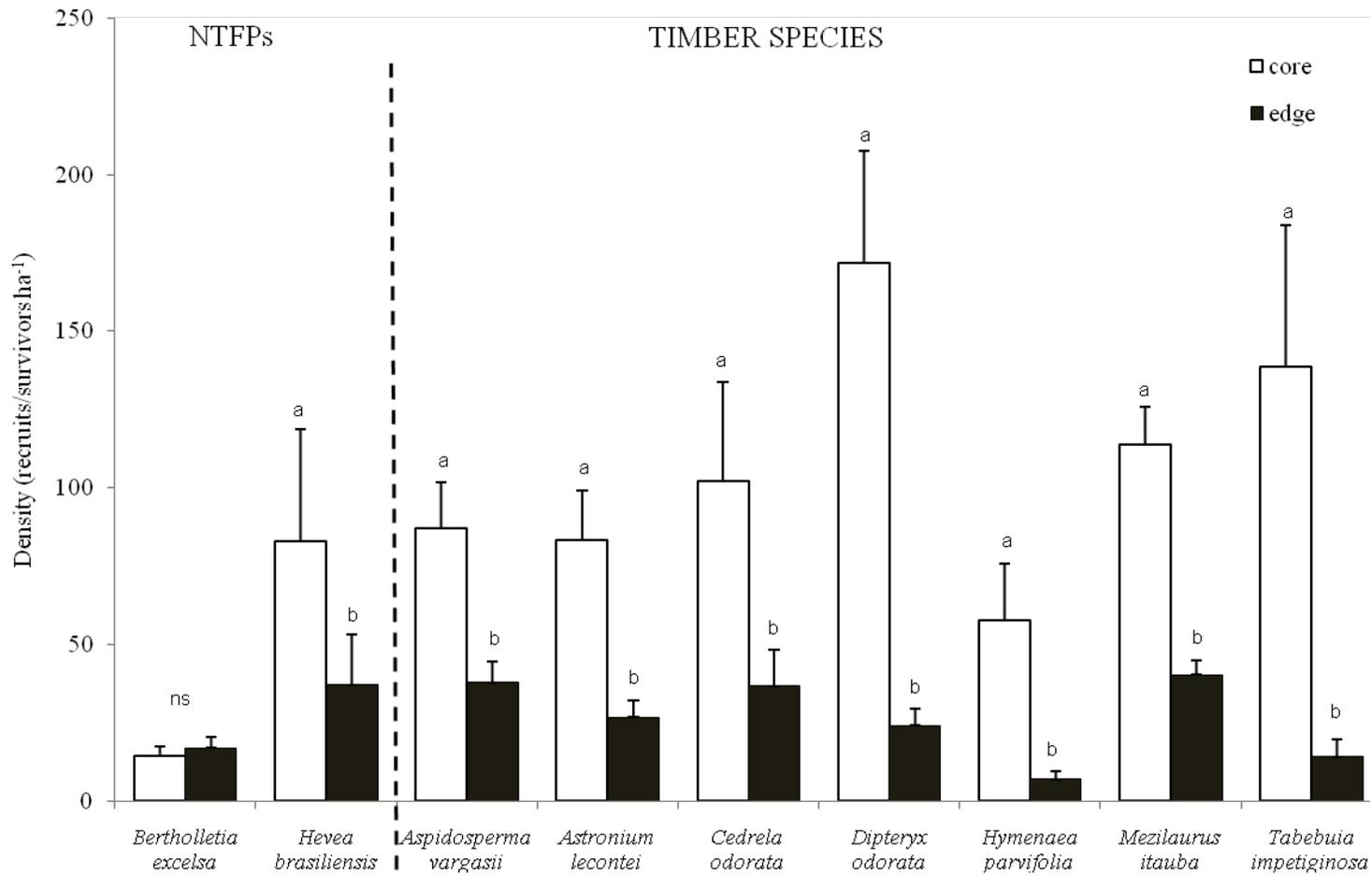


Figure 2-7. Density of individuals  $\leq 10$  cm dbh by microsite: plot core vs edge of disturbance. Different letters represent statistical differences at  $p \leq 0.05$ ; ns, no significant differences.

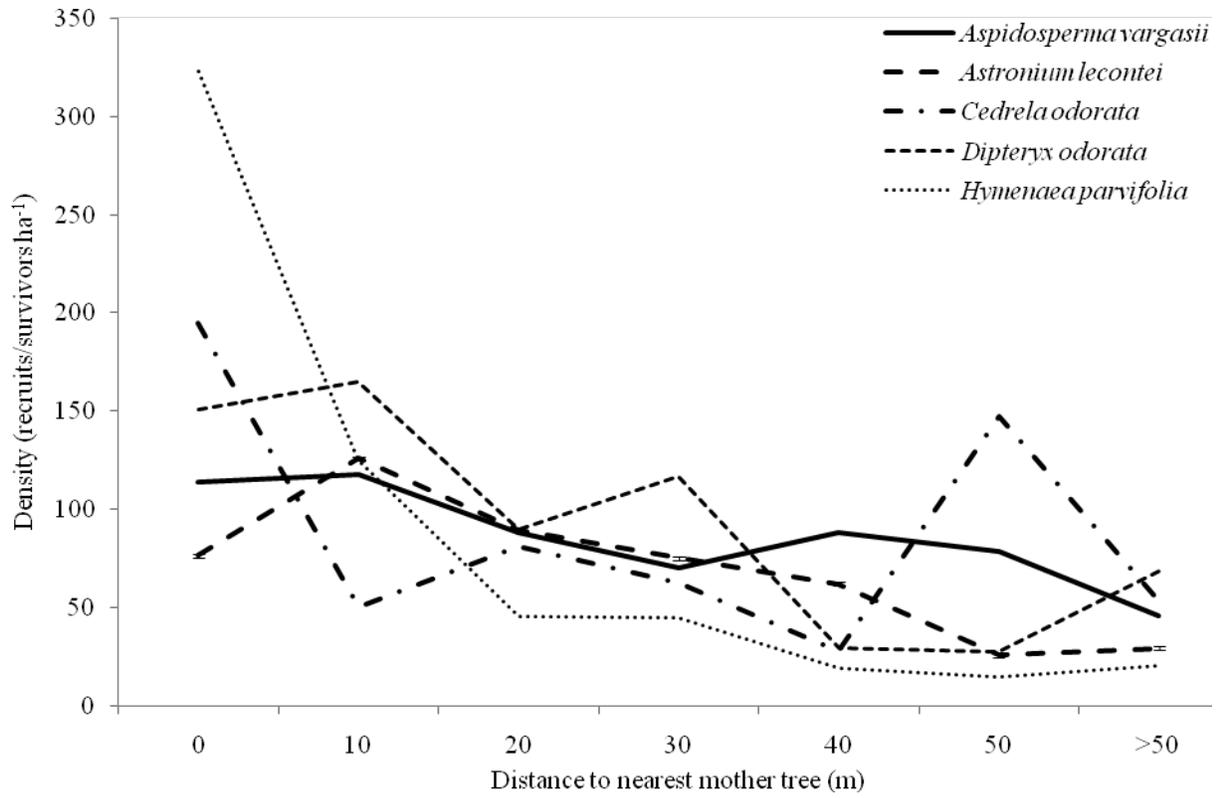


Figure 2-8. Significant effects ( $p \leq 0.05$ ) of distance to nearest potential mother tree on density of individuals  $\leq 10$  cm dbh of five timber species in Pando, Bolivia.

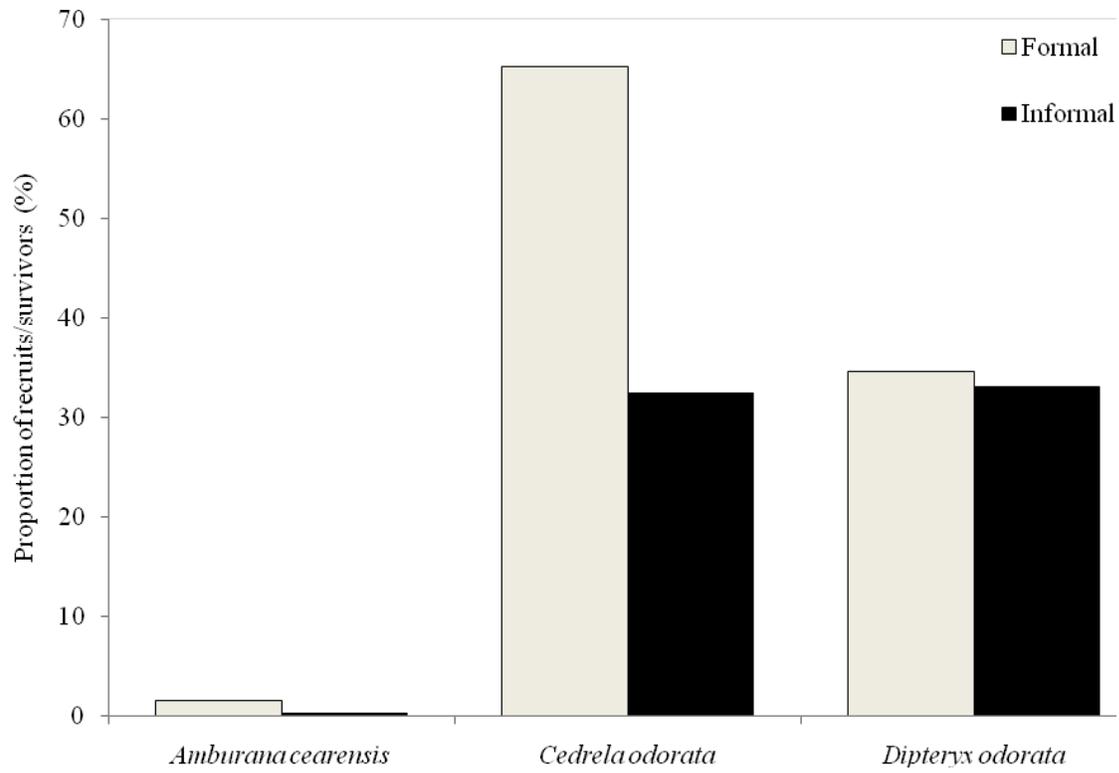


Figure 2-9. Proportion of conspecific individuals  $\leq 10$  cm dbh for species logged in both formal and informal logging sites.

## CHAPTER 3 CONCLUSIONS

Multiple use forest management (MFM) is a new concept that is becoming prominent worldwide (García-Fernández et al., 2008). Since timber exploitation started to take place in Northern Bolivia, unmanaged mixed forest use has simultaneously been occurring to the harvesting of non-timber forest products (NTFPs). MFM has been perceived as being especially important and relevant to the sustainability of these forests because *B. excelsa* has grown into a worldwide consumed tropical nut solely harvested from the wild. Some certification efforts to counteract potential effects of logging to *B. excelsa*'s population and/or nut production have taken place in this region. For example, certified landholdings are prohibited to harvest timber in either formal or informal ways (Duchelle, 2009). Brazil nut is particularly abundant in Northern Bolivia where hundreds of communities rely on its nuts, which are considered the most important source of cash income for the entire region, and have converted Bolivia into the largest exporter of *B. excelsa* in the world. The extent of dependency on *B. excelsa* nuts has proven to be highest in this region, followed by revenues from timber. A study of fourteen forest dependent community and social organizations assisted by an NGO for timber harvesting under management in lowland Bolivia indicated that 12% of family income in this region relies on non-timber forest products (mostly due to *B. excelsa* nuts) compared to only 0–2% in other forest dependent regions (Aguilar, 2008). Therefore, the income dependency by non-assisted communities on this species might be greater than those assisted communities. Furthermore, intensive cattle ranching and selective logging are acquiring growing interest in this region (Pacheco et al., 2009); as a result, pressure for non-sustainable usage of Brazil nut-rich forests is also meant to increase. If not appropriately tackled, current trends may have immeasurable consequences for these forests and people's livelihoods.

In this research we evaluated three different logging treatments (unlogged, formal and informal logging), and the extent these logging treatments are affecting natural regeneration (individuals  $\leq 10$  cm dbh) of economically important non-timber and timber species. We concluded that logging intensities are not different between unlogged, formal-, and informal-logging, but that informal logging, as practiced by community-based landholders, causes less forest disturbance than formal logging. While *B. excelsa* regeneration was unaffected by logging treatment or its location within a microsite zone (core or edge of disturbed areas), larger logging-caused disturbances (log landings vs skid trails) presented higher densities of this species. Regeneration densities from about half of examined species differed between logging treatments (N = 3 out of 8), logging disturbance (N = 4 out of 8), and all species preferred the core over the edge of disturbed areas. Somewhat worrisome is the low abundance of two of the most economically important timber species, *S. macrophylla* and *A. cearensis*. For these species, our analysis could not be properly run due to lack of occurrence in most sampled plots, an indication that these species urgently require silvicultural intervention to enhance their regeneration. Based on our results, it is clear that species specific management planning will be necessary to prevent lack of forest productivity of valuable species such as *C. odorata*, *T. impetiginosa*, and *D. odorata* over time. Although *B. excelsa* was the third species with the lowest abundance compared to the other examined species, the majority of *B. excelsa* regeneration has higher chances to survive over the long term (97% in fallows, and 100% in mature forests; Cotta et al., 2008).

Restricting forest use by establishing no-take areas within community lands might not be the only way for long term conservation of these forests. Results suggest that more sustainable efforts may be made to halt informal logging occurrence in these forests: by (1) encouraging low

intensity logging, (2) implementing combined traditional and scientific knowledge-based management activities, and (3) making incentives available to promote the utilization of silvicultural treatments among smallholders.

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## BIOGRAPHICAL SKETCH

Marlene Soriano was born in 1980, in Omereque, Campero province, Cochabamba, Bolivia. She grew up in the Valles Cruceños region in Santa Cruz. She received a Bachelor of Science degree in Forestry from Universidad Autónoma Gabriel René Moreno, Santa Cruz, Bolivia in 2006. She carried out her bachelor's thesis with the Bolivian Institute for Forestry Research; and since then, she has assisted and carried out research through various research projects along the different forest types in lowland Bolivia. Her research experience ranges from evaluating forest dynamics in mature forests, logged forests, and secondary forests, to estimating carbon for Reducing Emissions from Deforestation and Degradation (REDD) projects. She has collaborated with various types of local institutions, from researchers at Stanford University to forest dependent communities. In 2008, she enrolled in a graduate program at University of Florida in the School of Forest Resources and Conservation with a concentration in Tropical Conservation and Development. Her master thesis was conducted in collaboration with forest communities in Northern Bolivia where she previously worked and which greatly influenced her decision to pursue a graduate program. Her future plans are related to linking community conservation efforts to REDD programs by building capacity through participatory research.