

Provided for non-commercial research and education use.
Not for reproduction, distribution or commercial use.



This article appeared in a journal published by Elsevier. The attached copy is furnished to the author for internal non-commercial research and education use, including for instruction at the authors institution and sharing with colleagues.

Other uses, including reproduction and distribution, or selling or licensing copies, or posting to personal, institutional or third party websites are prohibited.

In most cases authors are permitted to post their version of the article (e.g. in Word or Tex form) to their personal website or institutional repository. Authors requiring further information regarding Elsevier's archiving and manuscript policies are encouraged to visit:

<http://www.elsevier.com/copyright>



Contents lists available at ScienceDirect

Forest Ecology and Management

journal homepage: www.elsevier.com/locate/foreco

Implementing multiple forest management in Brazil nut-rich community forests: Effects of logging on natural regeneration and forest disturbance

Marlene Soriano ^{a,b,*}, Karen A. Kainer ^{b,c}, Christina L. Staudhammer ^b, Eduardo Soriano ^d

^a Instituto Boliviano de Investigación Forestal, Casilla No. 6204, Santa Cruz, Bolivia

^b School of Forest Resources and Conservation, P.O. Box 110410, University of Florida, Gainesville, FL 32611, USA

^c Center for Latin American Studies, P.O. Box 115530, University of Florida, Gainesville, FL 32611, USA

^d Facultad de Ciencias Agrícolas, Universidad Autónoma Gabriel René Moreno, Santa Cruz, Bolivia

ARTICLE INFO

Article history:

Available online 14 June 2011

Keywords:

Bertholletia excelsa
Northern Bolivia
Formal logging
Informal logging
Regeneration density
Multiple-use

ABSTRACT

The encroachment of the logging frontier into Western Amazonia, where non-timber extraction has historically driven regional economies, provides an opportunity to explore the practice of multiple-use forest management. Families are now harvesting timber in their Brazil nut (*Bertholletia excelsa*)-rich community-titled forests, and we examined effects of formal and informal logging (with and without government-approved management plans, respectively) on forest disturbance and natural regeneration (individuals ≤ 10 cm dbh) of *B. excelsa* and 10 timber species in Northern Bolivia. *B. excelsa* regeneration densities 2–5 years after timber harvests did not differ between unlogged or logged (formally nor informally) sites; 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. Regeneration of the two most valuable timber species, *Swietenia macrophylla* and *Amburana cearensis*, rarely occurred, and only where conspecific trees were felled, implying that population recovery of selected species may need post-harvest silvicultural interventions. In contrast, two other high-value timber species (*Cedrela odorata* and *Dipteryx odorata*) responded favorably to disturbances. Contrary to our expectations and reported industrial-scale findings, formal logging resulted in a larger percentage, but still acceptable level, of disturbed area than informal logging ($10.6\% \pm 0.65$ SE vs $6.9\% \pm 1.26$ SE; $p = 0.047$).

Our overall finding that Brazil nut regeneration was unaffected by low logging intensities suggests that combined Brazil nut and timber harvests are compatible under certain circumstances. Still, adhering to legal requirements has been challenging for individual landholders. Preparing legally-required management plans without assistance is difficult, and individual landholders seek to harvest timber over multiple years, which is discouraged by formal logging that indirectly entails one-time extraction of nearly all commercial stems from the approximately 200 ha of forest landholders dedicate to timber production. Thus, reconciliation of legal requirements with community conditions is fundamental to long-term success of multiple-use forest management.

© 2012 Elsevier B.V. All rights reserved.

1. Introduction

Multiple-use forest management (MFM) that integrates management for diverse products such as timber, non-timber forest products (NTFPs), and environmental services, is being examined as an alternative that provides socioeconomic incentives to maintain standing tropical forests (Smith and Scherr, 2003; García-Fernández et al., 2008; Guariguata et al., 2010). The encroachment of the logging frontier into Western Amazonia, where the non-timber sector has historically driven regional economies, has captured the attention of conservationists and policy makers towards

* Corresponding author at: Instituto Boliviano de Investigación Forestal, Casilla No. 6204, Santa Cruz, Bolivia. Tel.: +591 3 341 1171; fax: +591 3 344 6453.

E-mail address: msoriano@ibifbolivia.org.bo (M. Soriano).

implementation of MFM that combines timber and non-timber products in this region (Guariguata et al., 2008). In recent decades, Brazil nut (*Bertholletia excelsa*: Lecythidaceae) has emerged not only as a key economic species for thousands of Peruvian, Brazilian and Bolivian forest dwellers and peri-urban processors (Stoian, 2005), but also a focus for conservation efforts, given that it is the only internationally-traded nut that comes strictly from mature tropical forests (Ortiz, 2002). And in Bolivia, this non-timber product has the distinction of being the single-most important exported forest species, constituting 39% (US\$ 73 of \$ 187 million) of the forest-based national economy (Cámara Forestal de Bolivia, 2007). However, just as Brazil nut exports are steadily rising in Bolivia, so are timber exports (Cámara Forestal de Bolivia, 2007), parallel to increased logging pressure on these Brazil nut-rich forests (Cronkleton et al., 2009).

Northern Bolivia serves as a nexus for these Brazil nut–timber interactions where 245 campesino and five indigenous communities have legal title to 3.5 million ha of forest (Pacheco et al., 2009). This level of community control over forest resources came on the heels of sweeping land tenure reforms initiated through the 1996 National Agrarian Reform Law. Further pressed by social movements, the national government issued decrees that recognized community rights over vast forested areas (Cronkleton et al., 2010). Although legal rights were granted at the community level, the actual size of the community polygon was determined by the number of families in the community, multiplied by a family allotment of approximately 500 ha (Cronkleton et al., 2010). Complementarily, the Forestry Law “Ley 1700” also passed in 1996 (Ley Forestal, 1996), recognized the legitimacy of traditional communities to timber rights for the first time (Pacheco, 1998). Since then, communities in Northern Bolivia have legal authority to engage in multiple use forest management through timber and non-timber extraction (MDS and VRNyMA, 2005; Superintendencia Forestal, 2008).

Most forest-use decisions in these communities, however, are made at the 500 ha family level, and many families are indeed opting to allow logging in their Brazil nut-rich forests (Guariguata et al., 2008). How compatible might these two activities be? Guariguata et al. (2008) highlight favorable complementarities such as a temporal separation of labor when harvesting Brazil nuts (rainy season) and timber (dry season), legal protection of *B. excelsa* from felling, and that inventories necessary for authorized timber harvest could simultaneously tally Brazil nut trees. Furthermore, Guariguata et al. (2009) determined that low-intensity logging (0.5 trees ha⁻¹) using nationally mandated reduced impact logging (RIL) guidelines, resulted in minimal damage to Brazil nut trees. Complementing adherence to RIL, additional legal responsibilities tied to timber extraction include developing a government-approved general forest management plan (GFMP) and annual operational logging plans (AOLPs). 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. Within this legal framework, a minimum 20-year harvest rotation is required to manage forest areas >200 ha, and one-time harvest plans are required for forest areas <200 ha (MDSyMA, 2000). Most logging companies, however, make contracts with individual community landholders (families) to harvest timber on areas ≥ 200 ha, an area considered large enough to make timber management profitable (Superintendencia Forestal, 2001). In either case, the GFMP must follow the minimum 20-year harvest rotation. Thus, logging is considered “legal” upon formal approval of these plans. Few communities and individual landholders, however, possess the capacity and capital to comply with these legal requirements (Martínez Montaña, 2008), and some hesitate to commit their entire 200 ha to timber production under a one-time logging plan. Thus, many enter into informal logging agreements with third parties who can be small-scale local loggers, legally established logging companies, or even community members. In contrast to the planning and harvest restrictions that characterize formal (legally-approved) logging with multiple timber species, families who enter into informal logging often dispense with a formal inventory and road planning, opting instead to open short paths to a few high-value trees. Loggers then use these paths to extract the timber, either in the form of *cuartones* (tree stems sliced into planks) or as logs veiled under legal documentation by a logging company from a separately approved AOLP with over-estimated volumes (Martínez Montaña, 2008; Superintendencia Forestal, 2007). Thus, while combining timber and Brazil nut harvests in a multiple-use forest management scheme seems promising, adhering to legal requirements has been a challenge. Yet, logging in Brazil nut-rich forests, is clearly on the rise as families

seek to reduce their dependency on Brazil nuts, a product whose fluctuating prices are determined by complex global markets. This logic seems sound given that a recent study of 131 Pando households found that Brazil nuts alone contributed 43% of the total family income (Duchelle et al., 2011).

In this study, we focus on the extent to which these escalating logging activities (both formal and informal timber harvests) are affecting Brazil nut populations of Northern Bolivia’s forest-based communities. Specifically, we empirically test the effects of formal and informal logging on densities of new recruits and survivors post-logging (individuals ≤10 cm dbh, hereafter referred as to regeneration) of Brazil nut and 10 timber species. Brazil nut is the central focus of this study because it plays the most important economic role for local economies and these 10 timber species were selected based on data from species logged from all research sites. We targeted this early stage in the life cycle of *B. excelsa* and other species because recruitment of a steady supply of new individuals into a population is basic to survival of that population and is the essence of sustainable forest use (Peters, 1996). Furthermore, to our knowledge, the effects of logging on Brazil nut regeneration have not yet been quantified. Because *B. excelsa* has been observed to respond favorably to increased light levels caused by larger disturbances, such as shifting cultivation (Cotta et al., 2008), and smaller ones, such as forest gaps (Myers et al., 2000; Zuidema and Boot, 2002), we hypothesized that logged areas (formal and informal) in general would contain enhanced *B. excelsa* regeneration densities when compared to unlogged forest. To more precisely assess the variation in habitats created by these two logging types, we also differentiated between skid trails, access roads, felling gaps, and log landings in both formally- and informally-logged sites. To further explain some of the mechanisms behind *B. excelsa* regeneration, we quantified the following variables within each disturbance type: core and edge zones of disturbed areas (hereafter referred to as microsite), distance to nearest conspecific potential mother tree, and canopy openness. Similar questions were also addressed for 10 commercially important timber species.

To further understand stand-level impacts induced by formal and informal logging, we also compared the extent of logged area, number of logged species, logging intensity, and percent of disturbed area to complement the regeneration information. By solely focusing on few high-value species, informal logging can quickly diminish the population of those target species by immediately extracting reproductive adults and undermining future regeneration (Schulze and Zweede, 2006). Because conspecific densities of most tropical species are extremely low (often <1 individual ha⁻¹) (Peters, 1996), this type of informal, selective logging enlarges the operational logging area needed to meet these species-specific market demands, resulting in lower logging intensities (Grogan et al., 2008). Some studies examining large-scale logging operations have demonstrated that RIL, such as required under formal logging in Bolivia, can reduce residual damage up to 50% compared to informal (or illegal) logging (Johns et al., 1996; Pinard and Putz, 1996; Sist et al., 1998). Therefore, we hypothesized that informal logging would cause greater disturbance, increase logging area, and thwart species abundance and diversity, but could also occur at lower logging intensities than formal logging within community forests in Northern Bolivia.

2. Methods

2.1. Study area

This study was implemented within community landholdings in the municipality of Filadelfia, located in the Western side of

the department of Pando in Northern Bolivia. Pando receives average annual rainfall of 1700–2000 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. There is high diversity of tree species, between 52 and 122 species ha⁻¹, and tree abundance, between 544 and 627 trees (≥ 10 cm dbh) ha⁻¹ (Mostacedo et al., 2006). Compared to neighboring states in Brazil and Peru, deforestation in Pando remains low because of its historical isolation from the rest of the country and its NTFP-rich forests (Pacheco, 1998; Duchelle, 2009). Pando communities began adopting timber management plans in 2005, and by 2006, approximately 16 communities had legally approved management plans (Benneker, 2008). While this upward trend to participate in formal logging continues with support from NGOs and in coordination with logging companies, many communities have begun to reject formal management on their lands for various reasons (e.g., expected benefits were not forthcoming, landholder perceptions that forests under formal management were not better off than those without, unfulfilled contracts, and challenging bureaucracies) (Benneker, 2006; Guarena and Borobobo, personal communication). Because of these constraints, informal logging has also made significant inroads in the province of Filadelfia. Within Bolivia, a large proportion of logs derived from informally-logged sites are combined with legally extracted timber to fulfill authorized timber volumes from managed areas (Pacheco et al., 2009). Moreover, because of its location proximate to Peru, logs also flow over the border and are subsumed in the Peruvian timber market. The existence of both formal and informal logging in Filadelfia's Brazil nut-rich forests makes it an appropriate site for addressing our research questions.

2.2. Field data collection

Fieldwork was conducted in six (two formally- and four informally-logged) Brazil nut-rich forest sites in three communities. Potential sites were preliminary selected based on research needs for formally- and informally-logged Brazil nut-rich forests. Subsequently, specific sites were chosen based on criteria established for time since timber harvest (Table 1) and landholders' willingness to collaborate. These latter two criteria narrowed down potential sites to those selected, which were considered representative of families and communities who have collected Brazil nuts over decades in Pando, and are now experimenting with timber harvests. Each logged site was spatially independent from the others, except for one site in which both, formally- and informally-logged sites overlapped, but effects of each logging type could be differentiated. Based on our field observations, densities of *B.*

excelsa were similar among sites. While nuts from all sites were collected annually, typically on a daily basis during the primary production season between January through March (Duchelle et al., 2011), timber harvesting only occurred once within each sampled area, 2–5 years prior to data collection. Differences in number of years since logging were balanced in both logging types, thus legitimizing comparisons across sites (see Table 1). Additionally, all but the most distant site were often visited for subsistence hunting.

All types of logging disturbances within logged sites 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 required by Bolivia's forestry law. In contrast, informal logging areas did not. To delineate these ill-defined areas, we used terrain observations and satellite imagery (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 noted that natural features such as wetlands, rivers, and/or slopes often defined 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 by less than 1% from the original predetermined area listed on the formal management plan. This finding confirmed that our border delineation method was realistic and dependable 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 7–10 plots in each disturbance type (skid trails, access roads, felling gaps, and log landings) within each formally- and informally-logged site; and 9–10 plots within unlogged areas adjacent to each logged site. In total, we established 203 plots allocated as follows in each disturbance type (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 (six formal, four informal), and 57 plots in unlogged areas (20 formal, 37 informal). Felling gaps were randomly selected from each site, with plot size varying with the extent of disturbance caused by the logged 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

Table 1
Stand characteristics of formal and informal logging sites.

Logging type	Logging site	Year of logging	Logging equipment	No. of logged species (SE)	Logged area ha (SE)	Logging intensity (SE)		% Disturbed area (SE)
						trees ha ⁻¹	m ³ ha ⁻¹	
Formal	A	2007	Skidder	7	225.6	0.28	1.6	11.2
	B	2004	Caterpillar	10	203.8	0.28	1.9	9.9
	Mean	2		8.5 (1.5)	214.7 (10.9)	0.28 (0.0)	1.8 (0.15)	10.6 (0.65)
Informal	A	2007	4-Wheeled drive truck	4	271.0	0.13	1.3	5.3
	B	2005	Skidder	5	153.6	0.25	1.2	7.2
	C	2007	Skidder	1	25.6	0.27	1.4	6.1
	D	2006	Skidder	3	32.0	0.31	2.5	9.1
Mean	4		3.3 (0.85)	120.5 (58.15)	0.24 (0.04)	1.6 (0.30)	6.9 (1.26)	
Pr (<0.05)	1-sided			0.0468*	0.1879	0.1845	0.4857	0.0468*

* Significant differences at $p \leq 0.05$, using the normal approximation for the two-sample Wilcoxon test.

installed in all log landings, with plot sizes varying with the extent of the landing. The 9–10 plots in unlogged forests were located at a random direction, typically from each sampled felling gap. Unlogged plots were 25 × 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. 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 approximately 2 (skid trails and access roads) to 3 m (felling gaps and log landings) into the forest. 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 logging-induced indirect sunlight penetrated into the forest edge. 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 (A1 and A2) at each side of the middle axis:

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

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

where: d = distance interval along the largest middle axis between two perpendicular distances, d_{left} = distance from the middle axis to the left side of disturbed area, d_{right} = distance from the middle axis to the right side of disturbed area, n = number of perpendicular lines extended every 5 m along the middle axis.

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

In each plot, we carefully searched and counted all regeneration of *B. excelsa* and 10 timber species. All examined timber species were harvested from study sites, only numbers of species harvested at each site were variable. We recorded microsite (core or edge) location of each individual. To assess whether distance to potential mother tree affected regeneration densities, we searched for potential mother trees up to 50 m from the border of the plot following Cotta et al.'s (2008) method for maximum *B. excelsa* seed dispersal. This distance was adopted for use with all species. For *B. excelsa*, distance to, and dbh of nearest potential mother trees (those ≥ 40 cm dbh) were recorded in all 203 plots whether regeneration of this species was encountered in the plot or not. The estimated potential mother tree size of ≥ 40 cm dbh for *B. excelsa* was adapted from Kainer et al. (2007). For all other species, distance to, and dbh of nearest potential mother trees were only recorded when individuals of that given species were found in the plot. Therefore, influence of mother tree for these other species was only estimated in the presence of regeneration of a given species. Finally, potential mother trees of these species were considered to be those ≥ 20 cm dbh, applying a standardized diameter that approximates

when most Amazonian tree species begin fruiting (Van Rheenen, 2005).

2.3. Data analysis

All statistical analyzes were performed using SAS software (Version 9.2), and differences were considered statistically significant at p -values ≤ 0.05 . Because the sample sizes to determine differences on extent of disturbance between logging treatments (two formal and four informal) were small, we used a two-sample non-parametric Wilcoxon test to test differences between logged areas, number of logged trees and species, logging intensities (trees ha^{-1} and $\text{m}^3 \text{ha}^{-1}$), and percent of disturbed area.

A generalized linear mixed model was estimated with the SAS procedure PROC GLIMMIX to predict regeneration densities by species as a function of logging type, distance to nearest potential mother tree, disturbance type, and microsite. Due to low numbers of occurrence of some species, however, the model could only be estimated for 8 of the 11 examined species (*B. excelsa* and seven timber species). The response variable "regeneration density" (number of recruits and survivors post logging) was characterized with 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, regeneration density of *Astronium lecontei* was truncated at 1000 individuals. Least squares means were also generated and adjusted using Scheffe's method to account for multiple comparisons between logging types, disturbance types, microsites, 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 *B. excelsa* regeneration densities ($N = 196$ plots, $N =$ sampling size). For analytical purposes, distances to nearest potential mother trees were classified in seven classes. A first class was set for trees encountered within the plot; five classes represented sequential 10 m intervals from the plot, and the sixth class was designated 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) on *B. excelsa* regeneration, and therefore excluded plots in unlogged sites ($N = 143$). Finally, Model 3 only used those plots where *B. excelsa* individuals were present ($N = 49$) to explore whether microsite helped explain *B. excelsa* densities (Table 2).

Similar methods and logic described above 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 were encountered in a particular plot, a more basic model (Model 0) was also estimated. Model 0 only accounted for two explanatory variables, logging treatments and canopy openness, to observe differences between logging treatments based on a larger sample size (195–196 plots, varying by species). 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. The ratio of the generalized chi-square statistic to its degrees of freedom was used to assess model fit. A value of this statistic close to one indicates that the variability in the data has been properly modeled, and that there is no residual overdispersion. For some of these species (*Cedrela odorata*, *Hymenaea parvilofia*, and *Mezilaurus itauba*), Model 3 showed slightly high chi-square/DF statistics, indicating poor model fit. Alternative models to Model 3, however, did not improve the statistics, and although residuals indicated no heteroscedasticity, results should be viewed conservatively. The effects of formal

Table 2
Best models to explain density of individuals ≤ 10 cm dbh of *B. excelsa* 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 unlogged plots to test whether densities differed between disturbance types (skid trails, access roads, felling gaps, log landings) and microsite (core and edge of disturbances). Model 3 included only plots where regeneration of the species under investigation was present.

Species	Model	No. of plots	Log-likelihood	Chi-square	p-Values for fixed effects				
					Canopy openness	Logging treatment	Disturbance type	Microsite	Distance mother
<i>NTPF species</i>									
<i>B. excelsa</i>	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
<i>Timber species</i>									
<i>A. 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>A. 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>C. 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>D. 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>H. 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>M. 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>T. 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*

* Significant differences at $p \leq 0.05$.

and informal logging on regeneration density of three conspecific logged species, as well as the relative abundance of examined species encountered in the 203 plots, were compared using descriptive statistics in Microsoft® Office Excel 2007.

3. Results

3.1. 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^{-1} , $p = 0.184$; 1.8 vs 1.6 m^3 ha^{-1} , $p = 0.486$), formal logging resulted in more species logged in a given logging site (61 ± 3.0 SE vs. 23 ± 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 1).

3.2. Explaining Brazil nut regeneration

Relative regeneration abundance (individuals ≤ 10 cm dbh) of *B. excelsa* was very low proportional to a pool of twelve species (0.67% of 100%) (Fig. 1). Regeneration densities did not differ between logging types (3.9 ± 1.3 ind. ha^{-1} in unlogged, 7.6 ± 1.7 ind. ha^{-1} in formally-, and 5.8 ± 1.2 ind. ha^{-1} in informally-logged sites, respectively; $p = 0.315$) (Fig. 2; Table 2) nor with distance to potential mother trees ($p = 0.477$) (Fig. 5; Table 2). Differences were, however, observed between disturbance types ($p = 0.027$)

(Table 2). Specifically, regeneration in log landings (15.1 ± 7.2 ind. ha^{-1}) was significantly higher than in skid trails (2.9 ± 1.2 ind. ha^{-1}) (Fig. 3). Differences between microsites (core versus edge of disturbed area) were observed for all species except for *B. excelsa* (14.6 ± 3.0 vs 17.1 ± 3.4 ind. ha^{-1} ; $p = 0.444$) (Fig. 4; Table 2).

3.3. Explaining regeneration of timber species

A. lecontei (30%), *C. odorata* (15%), and *M. itauba* (13%) were three of the most abundantly regenerating species found within all plots (unlogged, formally- and informally-logged) and across all types of disturbances (Fig. 1). In contrast, individuals of *Amburana cearensis* and *Swietenia macrophylla* were rare, representing only 0.04% and 0.21% of the 12 studied species, respectively (Fig. 1), and were only found in felling gaps where conspecific trees were felled.

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

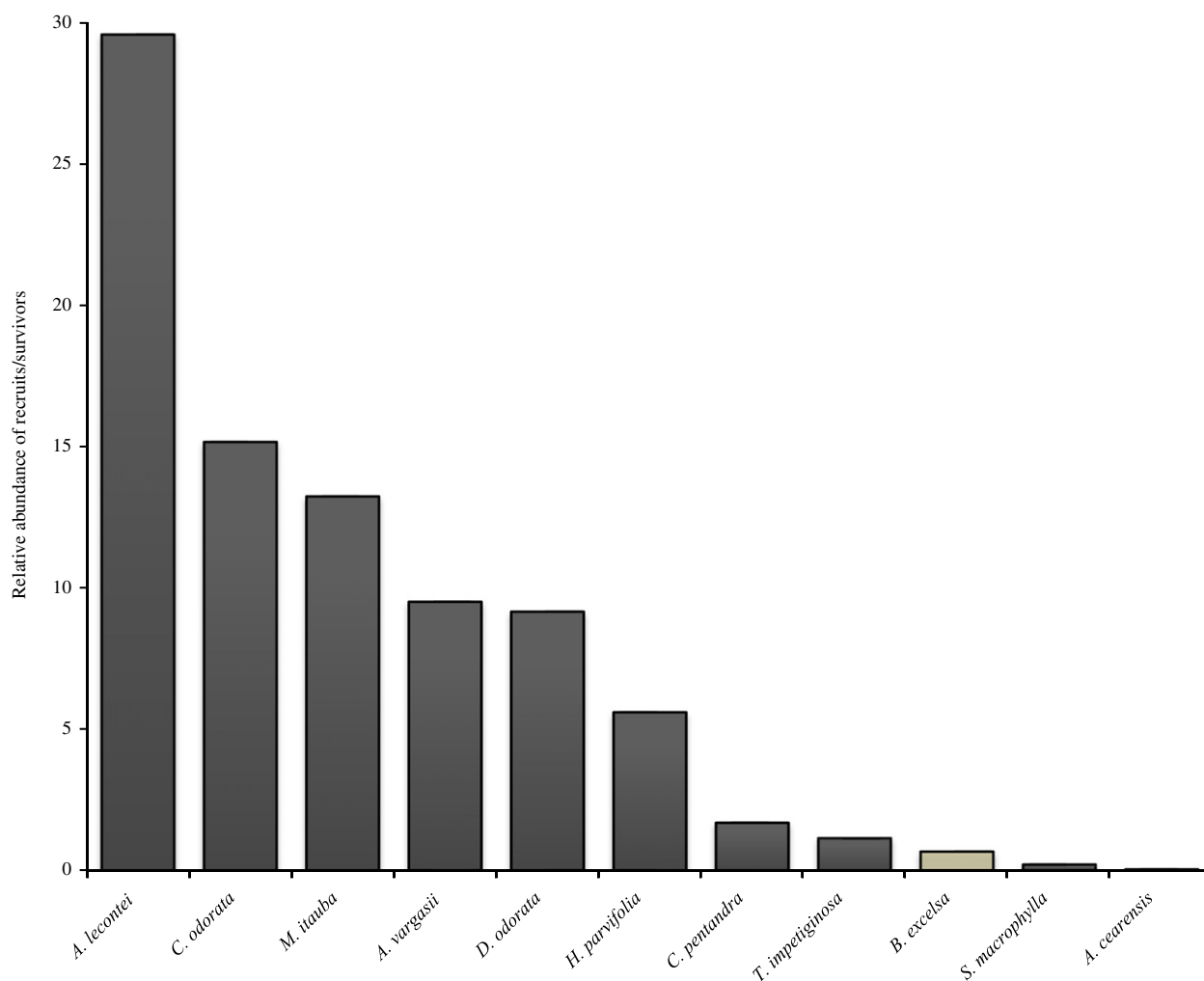


Fig. 1. Relative abundance of individuals ≤ 10 cm dbh in six logged sites and all types of logging disturbance by species. Timber species are represented by filled bars, and one non-timber forest products (NTFPs) species is represented by a light-colored bar. (For interpretation of the references to color in this figure legend, the reader is referred to the web version of this article.)

Some species demonstrated differences in regeneration densities between types of logging disturbances (Fig. 3; Table 2). *Aspidosperma vargasii* presented lower densities on skid trails (31.1 ± 8.4 ind. ha $^{-1}$) and access roads (15.7 ± 4.3 ind. ha $^{-1}$) than in felling gaps (84.2 ± 16.6 ind. ha $^{-1}$) ($p \leq .0001$). *D. odorata* had higher densities in log landings (198.1 ± 136.7 ind. ha $^{-1}$) than on skid trails (7.0 ± 3.1 ind. ha $^{-1}$), access roads (10.4 ± 3.9 ind. ha $^{-1}$), and felling gaps (17.1 ± 5.1 ind. ha $^{-1}$) ($p = 0.002$). Higher regeneration densities of *H. parvifolia* were, however, observed in felling gaps (24.2 ± 6.7 ind. ha $^{-1}$) and log landings (103.7 ± 70.7 ind. ha $^{-1}$) than on skid trails (2.7 ± 1.2 ind. ha $^{-1}$) and access roads (2.9 ± 1.3 ind. ha $^{-1}$) ($p \leq .0001$). Similarly, *M. itauba* also had higher densities in felling gaps (173.2 ± 23.2 ind. ha $^{-1}$) and log landings (227.8 ± 78.1 ind. ha $^{-1}$) than on skid trails (36.2 ± 6.9 ind. ha $^{-1}$) and access roads (40.2 ± 7.2 ind. ha $^{-1}$) ($p \leq .0001$).

Regarding regeneration microsite preferences, all species had higher densities at the core versus the edge of disturbed areas ($p \leq 0.05$) (Fig. 4; Table 2). 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 *Tabebuia imperiginosa* ($p = 0.001$) differed with distance to nearest potential mother tree (Table 2). Greater densities were observed at shorter distances, and then gradually decreased at greater distances (Fig. 5).

Individuals from the three most dominant species were identified within felling gaps to specifically determine whether 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 regardless of species logged (reaching up to 70% and 47% of all species, respectively). A high degree of conspecificity in regeneration establishment was observed where *C. odorata*, *D. odorata*, and *H. parvifolia* were logged: 41%, 34%, and 43% of all individuals were from these species, respectively. Only three species were logged in formal as well as in informal logging sites (*A. cearensis*, *C. odorata*, and *D. odorata*). Of these, only *C. odorata* showed differences, higher relative regeneration abundance in formally-logged than in informally-logged sites (Fig. 6).

4. Discussion

4.1. 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 ≤ 10 cm dbh) in abandoned agricultural fallows than in mature forests (Cotta et al.,

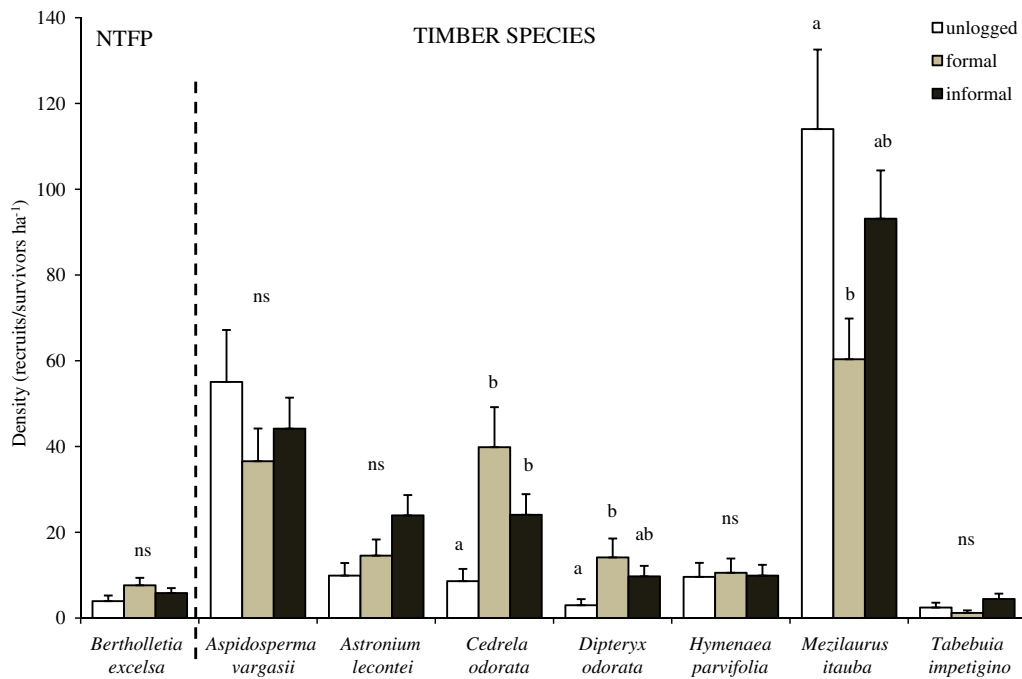


Fig. 2. Density of individuals ≤ 10 cm dbh in unlogged forest, and formally and informally logged sites by species. Different letters represent statistical differences at $p \leq 0.05$; ns, not significantly different.

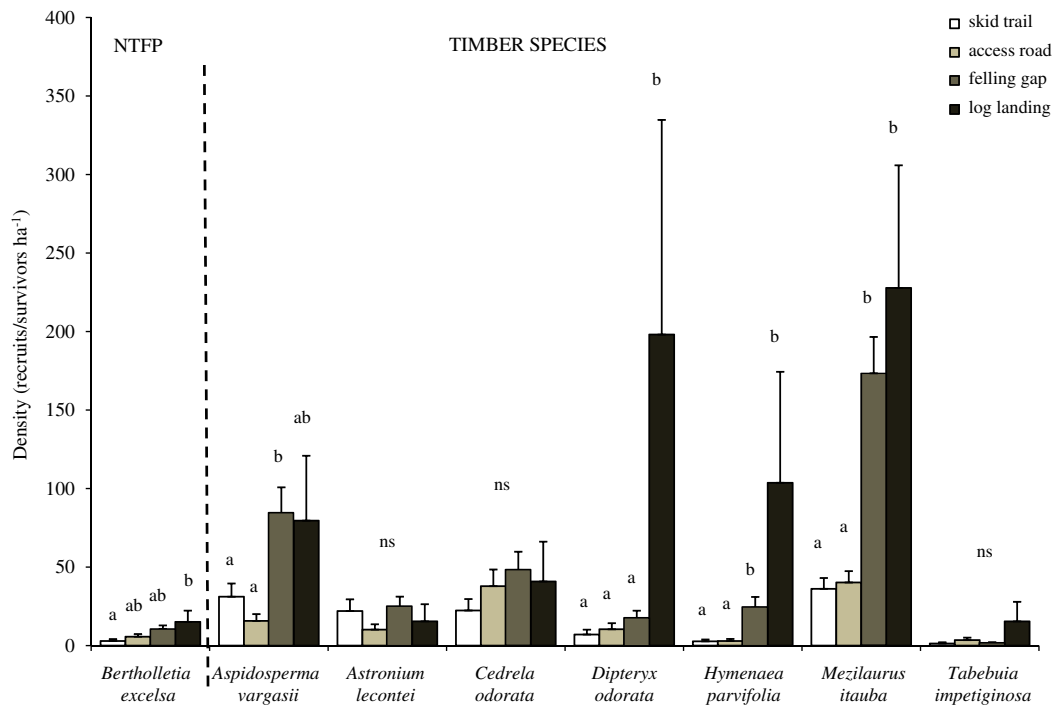


Fig. 3. Density of individuals ≤ 10 cm dbh by species and disturbance type. Different letters represent statistical differences at $p \leq 0.05$; ns, not significantly different.

2008), we found no statistical differences in *B. excelsa* regeneration densities between unlogged and logged (formal nor informal) sites. Though, densities of individuals ≤ 10 cm dbh in mature forests were slightly lower in our study sites than those of Cotta et al.'s (2008) in Acre, Brazil (3.9 vs 5.3 ind. ha⁻¹, respectively), densities in our logged sites (formal, 7.6; informal, 5.8 ind. ha⁻¹; Fig. 2) were much lower than in their abandoned agricultural fallows (12.7 ind. ha⁻¹). Still, we did observe higher densities in larger logging disturbances (i.e., log landings vs skid trails), corroborating our model in which canopy openness explained regeneration densities among

all logging treatments ($p \leq 0.001$) (Table 2). Observed regeneration density differences may reflect a combination of enhanced light and improved habitat for *B. excelsa* seed dispersers, although we did not measure the latter variable. Logging debris provides a measure of cover to the key disperser, agouti (*Dasyprocta spp.*), as it eats and buries *B. excelsa* seeds; opened fruits and recruits are often found within accumulated debris or behind tree stems (Peres and Baider, 1997). The lack of relationship between regeneration densities of individuals ≤ 10 cm dbh and distance to nearest potential mother trees contrasts with findings by Cotta et al. (2008) who

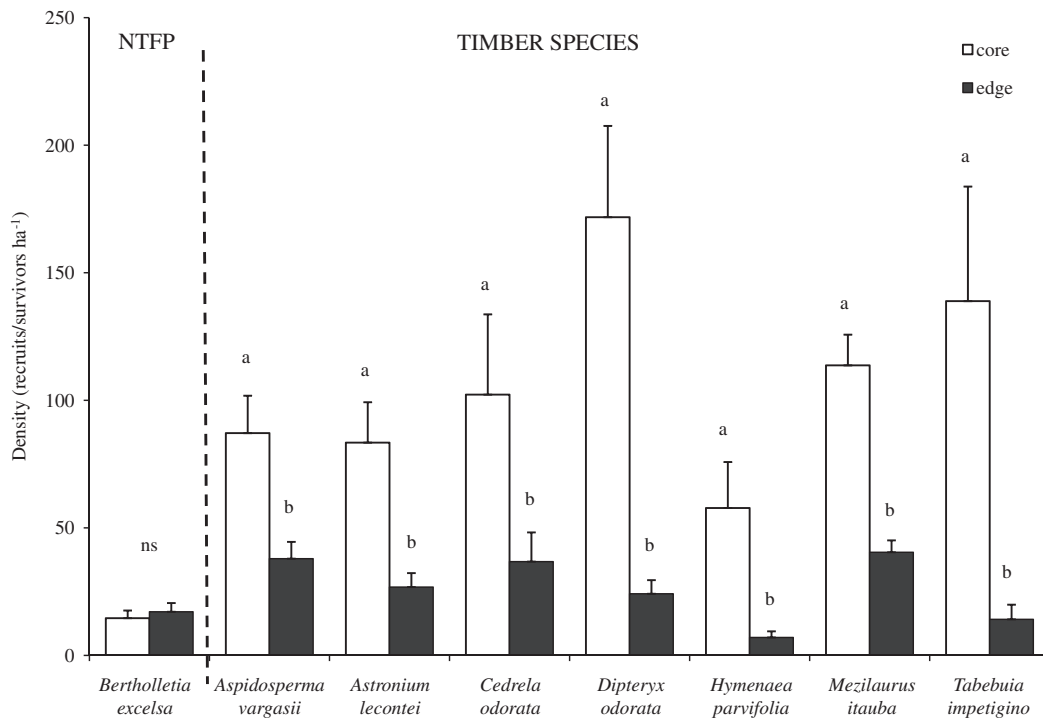


Fig. 4. Density of individuals ≤ 10 cm dbh by species and microsite (plot core vs edge of disturbance). Different letters represent statistical differences at $p \leq 0.05$; ns, not significantly different.

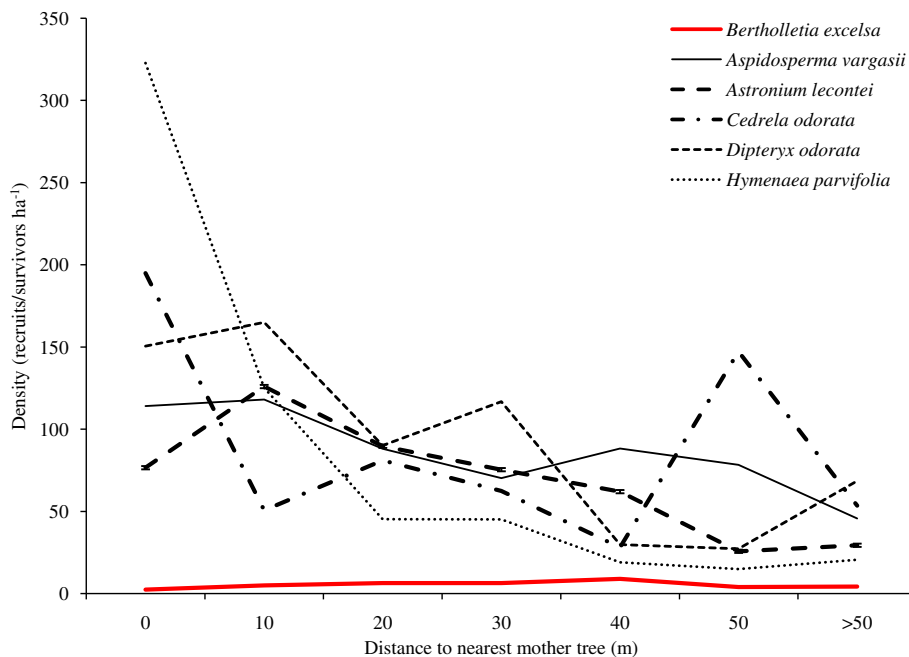


Fig. 5. Effect ($p \leq 0.05$) of distance to nearest potential mother tree on density of individuals ≤ 10 cm dbh of five timber species and Brazil nut in Pando, Bolivia.

found a positive correlation between proximity to mother trees and *B. excelsa* regeneration densities. One possible explanation for this comparative lack of effects in our Bolivian sites may be that these Brazilian study sites were originally used for shifting cultivation (suggesting sites that were pre-selected for low Brazil nut occurrence), thus making dependency on proximity to mother tree more noticeable. Alternatively, long distance seed reburial by agoutis also may have played an important role on seed dispersal (Tuck Haugaasen et al., 2010) and therefore seedling establishment. Our overall finding that neither formal nor informal logging

reduced regeneration of this keystone species under low logging intensities (Fig. 2; Table 2), complements Guariguata et al.'s (2009) results where low logging intensity plus application of RIL techniques caused minimal damage to *B. excelsa* trees ≥ 10 cm dbh in certified forest concessions.

4.2. The case of timber species

Logging under the conditions of our study sites resulted in greater regeneration of two important timber species; *D. odorata*

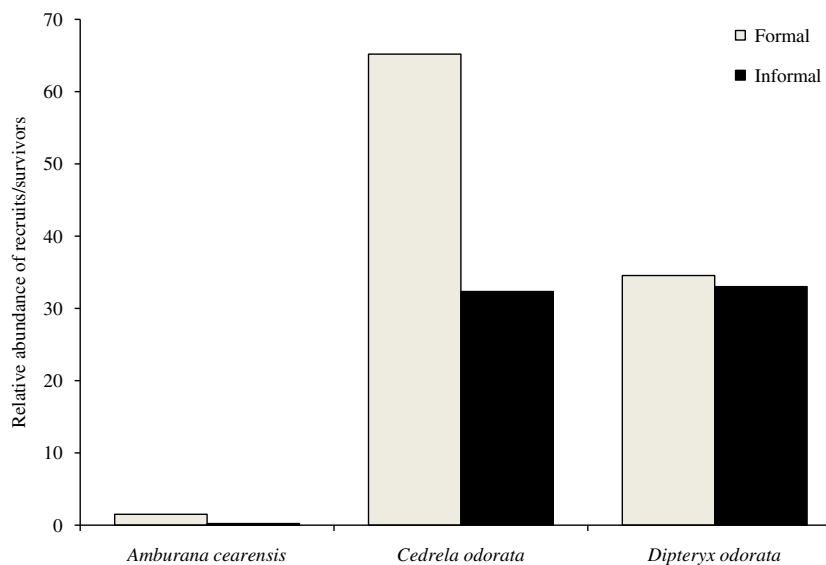


Fig. 6. Relative abundance of conspecific individuals ≤ 10 cm dbh for species logged in both formal and informal logging sites.

and *C. odorata* had higher densities in logged areas than in unlogged ones. 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 favorably 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 examined timber species (Table 2). 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 the logging season in Northern Bolivia. This finding 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*, are rarely encountered in the region (Mostacedo et al., 2009) as well as in our study plots – each species representing less than 1% of total counted individuals from all species (Fig. 1) – 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).

4.3. Disturbance created by formal and informal logging

Our hypothesis that the extent of logging disturbance would be greater in informally (illegally) logged sites than in those with legally-approved management plans (formally logged), as documented in multiple studies (Johns et al., 1996; Pereira et al., 2002; Pinard and Putz, 1996; Sist et al., 1998) was not supported. Contrary to our expectations, and despite statistically similar logging intensities (0.24 and 0.28 trees ha^{-1}), we found that informally (illegally) logged sites incurred a lower percentage of disturbed area ($6.9 \pm 1.3\%$) than formally logged sites ($10.6 \pm 0.6\%$). Our opposing results to these previous studies could be attributed to the fact that they were carried out within large (industrial) scale logging

operations, while logging activities in our community landholding sites were relatively small-scale. We ascribe the observed comparatively lower extent of disturbance in our informally (vs formally) logged sites to two causes: 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). These types of expert local knowledge and interventions point to the benefits of integrating local peoples into forest management planning and practice, particularly in community settings where forest values are diverse and often not understood by loggers and/or professional forest managers (Guariguata et al., 2008). Secondly, though not statistically different, we measured greater logging intensities and larger numbers of logged trees in formally- than in informally-logged sites (Table 1), technical variables that could also impact extent of forest disturbance. Finally, not all legally-approved timber harvestings are alike. Guariguata et al. (2009) found that Forest Stewardship Council-certified logging on three large ($\sim 40,000$ – $154,000$ ha) timber concessions with harvest experience in Brazil nut-rich Pando forests had lower values for percent area disturbed (5.3–6.1%) than in our formally-logged sites (9.9–11.2%). Although our measure of extent of disturbance also included forest edges (2–3 m around disturbed areas) and not only the area of disturbed canopy as realized by Guariguata et al. (2009), their values would still be comparatively lower than ours. Their values were, however, quite similar to our informally-logged sites (5.3–9.1%), particularly when considering methodological differences in disturbance measurements. Their results were attained, however, despite slightly higher timber harvest intensities (0.5–0.6 trees ha^{-1} compared to 0.1–0.3 trees ha^{-1} found in our study), suggesting that with practice and under the rigorous hand of forest certification, logging operations can generate minimal forest disturbance in Northern Bolivia. Certified smaller-scale operations, in particular, could potentially minimize forest disturbance to even lower values than what are presented herein with further implications for reducing carbon emissions.

4.4. Is multiple-use forest management viable in Northern Bolivia's communities?

Our overall finding that *B. excelsa* regeneration (≤ 10 cm dbh) was unaffected by logging at low intensities complements

Guariguata et al.'s (2009) results of minimal damage to *B. excelsa* trees ≥ 10 cm dbh at low logging intensities and under RIL guidelines. These collective results suggest that combined Brazil nut and timber harvests (a common form of multiple forest use in Northern Bolivia) are compatible under certain biophysical and organizational circumstances. A key enabling condition was that timber in our study sites was harvested at low intensities (0.13–0.31 trees ha⁻¹), falling within the lowest range of logging intensities found in the literature (Johns et al., 1996; Sist et al., 1998; Rockwell et al., 2007), and resulting in acceptable levels of forest disturbance. Furthermore, our findings of statistically higher levels of *B. excelsa* regeneration in log landings versus smaller disturbances, such as skid trails, suggests regeneration success of this key NTFP even at the most disturbed portions of the logged landscape. Additionally, small-scale operations (7–10 trees per family parcel), allowed greater control over the logging operation by local landholders. Indeed, perhaps the most critical factor contributing to minimal disturbance by logging operations in these Brazil nut-rich forests was the effective participation of resident landholders, particularly where informal logging with no universally-accepted logging practices occurred. Local residents hand-picked targeted harvest trees, identified the best route to minimize collateral damage from timber removal, and remained vigilant of harvest operations at the time of logging. This type of community and individual landholder vigilance could also mitigate potential problems under formal management plans such as continued and expanded use of logging roads by informal loggers and wildlife poachers to access remote areas that were otherwise inaccessible (Asner et al., 2006; Fredericksen, 2000). These examples of local knowledge and supervision just begin to scratch the surface of the synergies that can occur when effectively partnering with local people to improve MFM research and applications. Guariguata et al. (2010) identify local knowledge as a key factor in advancing compatibility between timber and non-timber systems, and research partnerships with local stakeholders can increase research relevance, enhance knowledge exchange, and result in greater research and implementation success (Kainer et al., 2009). Doing science together (Fortmann, 2008) seems a necessary step when exploring new ways to manage complex ecological systems for socioeconomic benefits. Thus, based on our field assessments of regeneration, both formal logging with approved management plans as well as informal logging conducted under the supervision of individual landholders, seem technically compatible with Brazil nut harvests. Still, multiple challenges to both logging systems within these Brazil nut-rich forests remain.

4.5. In the shoes of community-based landholders

To be profitable, formal logging typically requires extraction of nearly all commercial stems from the approximately 200 ha of productive forest in a single year (Cronkleton et al., 2009), indirectly prohibiting small-scale timber production. This one-time harvest means that individual families receive a large payment in one lump sum, which can be more challenging to manage than income generation spread over several years into the future. Furthermore, requirements to prepare GFMPs are nearly impossible to abide by without technical assistance and economic support of external institutions (Benneker, 2006; Martínez Montaña, 2008; Cronkleton et al., 2009). As a result, though not comfortable with participating in illegal activities, most community-based landholders in our study region tended to engage in informal logging. Their main reasoning was that over the long-term, formal logging could not guarantee a steady provision of economic benefits, nor could it improve the post-logging conditions of their forests as declared by promoters. Additionally, not all felled trees under formal logging make it to the market, translating into substantial economic losses for

smallholders because they do not even receive the expected logging income from those trees that could otherwise serve as future crop trees (Benneker, 2006; Paz and Borobobo, personal communication). Thus, these landholders came to the same conclusion as Martínez Montaña (2008) that under current conditions, formal logging was typically worse for their income and forests than illegally selling smaller quantities of high-value, informally-logged timber. Certainly, illegal logging has also clear detrimental ecological effects (Uhl and Vieira, 1989; Kometter et al., 2004). It solely targets high-value species, quickly diminishing these species from forest ecosystems (Fig. 1). Perhaps the most prejudicial fact is that illegally harvested tropical timber floods local timber markets, leading to devaluation of formally-logged timber (Kaimowitz, 2007). Therefore, it is no surprise that the impact of illegal timber harvesting in Bolivia accounts for an estimated, and economically devastating, 40% of the timber market (Andaluz and Mancilla, 2006).

4.6. Is there a pathway that fits those shoes?

The hope to sustainably integrate timber harvests into established Brazil nut extraction regimes for multiple-use management has been the focus of discussion over the last decade in Bolivia. Guidelines for commercial harvests of Brazil nut alone began with “Brazil nut management norms” (MDS and VRNyMA, 2005), but these fall short by ignoring the traditional system in place and imposing severe opportunity costs (Guariguata et al., 2008). A recent resolution to encourage integral forest management plans (PGIBs, Spanish acronym) (Superintendencia Forestal, 2008) begins to address both timber and non-timber management jointly. Still, while Brazil nut management plans are slowly being implemented supported by NGOs, no PGIB plans have been approved to date (Suarez, ex-Director of Pando's regional regulatory Authority-ABT, personal communication). Moreover, none of these legal guidelines addresses the most problematic issues in the region: illegal timber harvestings and the disconnect between property rights delineated at the community level and decision-making at the family level. We are certain, however, that it is possible to sustain these forests under a multiple use system given the inherent diversity of non-timber and timber products (i.e., Brazil nut, rubber, and multiple palm species, Mostacedo et al., 2009). Thus, development of reduced impact harvesting (RIH) techniques and ecological guidelines on harvestable volumes, percent of seed trees retention, and sound harvest rotations to allow population recovery of harvested species become central to PGIB. We also assert that, in Northern Bolivia, multiple-year small-scale timber harvesting by individual community landholders has better prospects than one-time harvests every 20 years as indirectly induced by formal logging. Developing simplified and integral legal procedures and guidelines (Martínez Montaña, 2008) that incorporate these community-level distinctions rather than ignoring them or integrating them as an afterthought (Schmink, 2004) paves the way for multiple-use forest management that better fits the shoes of community-based landholders.

Acknowledgments

This research would have not been possible without the support of the Amazon Conservation Leadership Initiative (ACLI), a University of Florida (UF) program funded by the Gordon and Betty Moore Foundation and matching support from UF's IFAS. We are thankful to the Centro de Investigación y Preservación de la Amazonia (CIPA) for their logistical support while in Pando. We thank Eben Broadbent, Jack Putz, and Bonifacio Mostacedo for their helpful comments and revisions of earlier drafts. Above all, support and interest expressed by community people from Comunidades Campesinas San Antonio, Purísima and Holanda, especially of the individual community landholders where this research took place were crucial for the

successful completion of data collection and this material. Discussion of results with representatives of sampled communities and local institutions in Pando that work with communities greatly enriched our final conclusions.

References

- Andaluz, A.W., Mancilla, R.T., 2006. Gobernabilidad y cumplimiento de la legislación en el sector forestal de Bolivia. The World Bank, Washington, DC, p. 62.
- Asner, G.P., Broadbent, E.N., Oliveira, P.J.C., Keller, M., Knapp, D.E., Silva, J.N.M., 2006. Conditions and fate of logged forests in the Brazilian Amazon. *Science* 103, 12947–12950.
- Benneker, C., 2006. From policy to practice: the development of a community forest enterprise – a case study from Bolivia. In: FT/RR1 (Eds.), *Community-based forest enterprises in tropical forest countries: scoping study*. International Tropical Timber Organization, Forest Trends, Rights and Resources.
- Benneker, C., 2008. Dealing with the state, the market and NGOs: the impact of institutions on the constitution and performance of Community Forest Enterprises (CFE) in the lowlands of Bolivia. PhD Thesis. Wageningen University, the Netherlands.
- Cámara Forestal de Bolivia, 2007. Anuario estadístico del sector forestal de Bolivia 2006. Cámara Forestal de Bolivia, Santa Cruz, Bolivia.
- Contreras, F., Cordero, W., Fredericksen, T.S., 2001. Evaluación del aprovechamiento forestal. Proyecto BOLFOR, Santa Cruz, Bolivia.
- Cotta, J.N., Kainer, K.A., Wadt, L.H.O., Staudhammer, C.L., 2008. Shifting cultivation effects on Brazil nut (*Bertholletia excelsa*) regeneration. *Forest Ecology and Management* 256, 28–35.
- Cronkleton, P., Pacheco, P., Iburgüen, R., Albornoz, M.A., 2009. Reformas en la tenencia forestal en Bolivia: la gestión comunal en las tierras bajas. CIFOR/CEDLA, La Paz, Bolivia.
- Cronkleton, P., Albornoz, M.A., Barnes, G., Evans, K., Jong, W., 2010. Social geomatics: participatory forest mapping to mediate resource conflict in the Bolivian Amazon. *Human Ecology* 38, 65–76.
- Duchelle, A.E., 2009. Conservation and livelihood development in Brazil nut-producing communities in a tri-national Amazonian frontier. Dissertation. University of Florida, USA.
- Duchelle, A.E., Cronkleton, P., Kainer, K.A., Guanacoma, G., Gezan, S., 2011. Resource theft in tropical forest communities: implications for non-timber management, livelihoods, and conservation. *Ecology and Society* 16, 4.
- Fortmann, L. (Ed.), 2008. *Participatory Research in Conservation and Rural Livelihoods: Doing Science Together*. Wiley-Blackwell, West Sussex, UK.
- Fredericksen, T.S., 2000. Aprovechamiento Forestal y Conservación de los bosques tropicales. Documento técnico. Proyecto BOLFOR, Santa Cruz, Bolivia.
- Fredericksen, T.S., Pariona, W., 2002. Effect of skidder disturbance on commercial tree regeneration in logging gaps in a Bolivian tropical forest. *Forest Ecology and Management* 171, 223–230.
- García-Fernández, C., Ruiz-Pérez, M., Wunder, S., 2008. Is multiple-use forest management widely implementable in the tropics? *Forest Ecology and Management* 256, 1468–1476.
- Grogan, J., Jennings, S.B., Landis, R.M., Schulze, M., Baima, A.M.V., Lopes, J.C.A., Morghauer, J.M., Oliveira, L.R., Pantoja, F., Pinto, D., Silva, J.N.M., Vidal, E., Zimmerman, B.L., 2008. What loggers leave behind: impacts on big-leaf mahogany (*Swietenia macrophylla*) commercial populations and potential for post-logging recovery in the Brazilian Amazon. *Forest Ecology and Management* 255, 269–281.
- Guariguata, M.R., Cronkleton, P., Shanley, P., Taylor, P.L., 2008. The compatibility of timber and non-timber forest product extraction and management. *Forest Ecology and Management* 256, 1477–1481.
- Guariguata, M.R., Licona, J.C., Mostacedo, B., Cronkleton, P., 2009. Damage to Brazil nut trees (*Bertholletia excelsa*) during selective timber harvesting in Northern Bolivia. *Forest Ecology and Management* 258, 788–793.
- Guariguata, M.R., García-Fernández, C., Sheil, D., Nasi, R., Herrero-Jáuregui, C., Cronkleton, P., Ingram, V., 2010. Compatibility of timber and non-timber forest product management in natural tropical forests: perspectives, challenges, and opportunities. *Forest Ecology and Management* 259, 237–245.
- Johns, J.S., Barreto, P., Uhl, C., 1996. Logging damage during planned and unplanned logging operations in the Eastern Amazon. *Forest Ecology and Management* 89, 59–77.
- Kaimowitz, D., 2007. Forest law, enforcement, and rural livelihoods. In: Tacconi, L. (Ed.), *Illegal Logging*. Earthscan, London, UK, pp. 110–138.
- Kainer, K.A., Wadt, L.H.O., Staudhammer, C.L., 2007. Explaining variation in Brazil nut fruit production. *Forest Ecology and Management* 250, 244–255.
- Kainer, K.A., DiGiano, M.L., Duchelle, A.E., Wadt, L.H.O., Bruna, E., Dain, J., 2009. Partnering for greater success: local stakeholders and research in tropical biology and conservation. *Biotropica* 41, 555–562.
- Kometter, R.F., Martinez, M., Blundell, A.G., Gullison, R.E., Steininger, M.K., Rice, R.E., 2004. Impacts of unsustainable mahogany logging in Bolivia and Peru. *Ecology and Society* 9, 12.
- Lemon, P.E., 1957. A new instrument for measuring forest overstory density. *Journal of Forestry* 55, 667–668.
- Ley Forestal, 1996. Ley No. 1700 del 12 de Julio de 1996. La Paz, Bolivia.
- Licona Vasquez, J.C., Peña Claros, M., Mostacedo, B., 2007. Composición florística, estructura y dinámica de un bosque Amazónico aprovechado bajo diferentes intensidades en Pando, Bolivia. Proyecto BOLFOR/Instituto Boliviano de Investigación Forestal, Santa Cruz, Bolivia.
- Martínez Montaña, J.A., 2008. Marco legal para el manejo forestal por pequeños productores y comunidades en las tierras bajas de Bolivia. CIFOR, FORLIVE Project.
- MDS (Ministerio de Desarrollo Sostenible) and VRNyMA (Viceministerio de Recursos Naturales y Medio Ambiente), 2005. Norma técnica para la elaboración de planes de manejo de castaña (*Bertholletia excelsa* Humb and Bonpl.). Resolución Ministerial 077/2005. La Paz, Bolivia.
- MDSyMA (Ministerio de Desarrollo Sostenible y Medio Ambiente), 2000. Normas técnicas forestales. DIREMA-UMAREMA, Prefectura de Santa Cruz, Bolivia.
- Mostacedo, B., Pinard, M., 2001. Ecología de semillas y plántulas de árboles maderables en bosques tropicales de Bolivia. In: Mostacedo, B., Fredericksen, T.S. (Eds.), *Regeneración y silvicultura de bosques tropicales en Bolivia*. Proyecto BOLFOR, Santa Cruz, Bolivia.
- Mostacedo, B., Balcasar, J., Montero, J.C., 2006. Tipos de bosque, diversidad y composición florística en la Amazonia sudeste de Bolivia. *Ecología en Bolivia* 41, 99–116.
- Mostacedo, B., Villegas, Z., Licona, J.C., Alarcón, A., Villarroel, D., Peña-Claros, M., Fredericksen, T.S., 2009. Ecología y silvicultura de los principales bosques tropicales de Bolivia. IBIF, Santa Cruz, Bolivia.
- Myers, G.P., Newton, A.C., Melgarejo, O., 2000. The influence of canopy gap size on natural regeneration of Brazil nut (*Bertholletia excelsa*) in Bolivia. *Forest Ecology and Management* 127, 119–128.
- Ortiz, E.G., 2002. Brazil nut (*Bertholletia excelsa*). In: Shanley, P., Laird, S.A., Guillén, A. (Eds.), *Tapping the Green Market: Certification and Management of Non-timber Forest Products*. Earthscan Publications Ltd., London, UK, pp. 61–74.
- Pacheco, P., 1998. Estilos de Desarrollo y su Impacto en la Deforestación y Degradación de los Bosques de las Tierras Bajas de Bolivia. Serie Bosques y Sociedad, 2. CIFOR, Centro de Estudios para el Desarrollo Laboral y Agrario, Taller de Iniciativas en Estudios Rurales y Reforma Agraria.
- Pacheco, P., Ormachea, E., Cronkleton, P., Albornoz, M.A., Paye, L., 2009. Trayectorias y tendencias de la economía forestal extractiva en el norte amazónico de Bolivia. CIFOR/CEDLA/RR1, La Paz, Bolivia.
- Peña-Claros, M., Peters, E.M., Justiniano, M.J., Bongers, F., Blate, G.M., Fredericksen, T.S., Putz, F.E., 2008. Regeneration of commercial tree species following silvicultural treatments in a moist tropical forest. *Forest Ecology and Management* 255, 1283–1293.
- Pereira Jr, R., Zweede, J., Asner, G.P., Keller, M., 2002. Forest canopy damage and recovery in reduced-impact and conventional selective logging in Eastern Para, Brazil. *Forest Ecology and Management* 168, 77–89.
- Peres, C.A., Baider, C., 1997. Seed dispersal, spatial distribution and population structure of Brazil nut trees (*Bertholletia excelsa*) in Southeastern Amazonia. *Journal of Tropical Ecology* 13, 595–616.
- Peters, C.M., 1996. The ecology and management of non-timber forest resources. World Bank Technical Paper No. 322. The World Bank, Washington, DC.
- Pinard, M.A., Putz, F.E., 1996. Retaining forest biomass by reducing logging damage. *Biotropica* 28, 278–295.
- Putz, F.E., Blate, G.M., Redford, K.H., Fimbel, R., Robinson, J., 2001. Tropical forest management and conservation of biodiversity: an overview. *Conservation Biology* 15, 7–20.
- Rockwell, C.A., Kainer, K.A., Staudhammer, C.L., Baroloto, C., 2007. Future crop tree damage in a certified community forest in Southwestern Amazonia. *Forest Ecology and Management* 242, 108–118.
- Schmink, M., 2004. Communities, forests, markets, and conservation. In: Zarin, D.J., Alavalapati, J.R.R., Putz, F.E., Schmink, M. (Eds.), *Working Forests in the Tropics: Conservation Through Sustainable Use*. Columbia University Press, New York, pp. 119–129.
- Schulze, M., Zweede, J., 2006. Canopy dynamics in unlogged and logged forest stands in the Eastern Amazon. *Forest Ecology and Management* 236, 56–64.
- Sist, P., Nolan, T., Bertault, J.G., Dykstra, D., 1998. Harvesting intensity versus sustainability in Indonesia. *Forest Ecology and Management* 108, 251–260.
- Smith, J., Scherr, J.S., 2003. Capturing the value of forest carbon for local livelihoods. *World Development* 31, 2143–2160.
- Soriano Candia, M., 2005. Caracterización reproductiva y germinativa de especies forestales maderables en bosque seco Chiquitano en Santa Cruz, Bolivia. B.Sc. thesis. Universidad Autónoma Gabriel René Moreno, Santa Cruz, Bolivia.
- Stoian, D., 2005. Making the best of two worlds: rural and peri-urban livelihood options sustained by nontimber forest products from the Bolivian Amazon. *World Development* 33, 1473–1490.
- Superintendencia Forestal, 2001. Resoluciones, directrices, e instructivos emitidos por la Superintendencia Forestal. Instructivo No. 003/2001. Santa Cruz, Bolivia.
- Superintendencia Forestal, 2007. Informe Anual 2006. Sistema de Regulación de Recursos Naturales Renovables, Santa Cruz, Bolivia.
- Superintendencia Forestal, 2008. Resolución No. 174/2008. Santa Cruz, Bolivia.
- Tuck Haugaasen, J.M., Haugaasen, T., Peres, C.A., Gribel, R., Wegge, P., 2010. Seed dispersal of the Brazil nut tree (*Bertholletia excelsa*) by scatter-hoarding rodents in a central Amazonian forest. *Journal of Tropical Ecology* 26, 251–262.
- Uhl, C., Vieira, I.C.G., 1989. Ecological impacts of selective logging in the Brazilian Amazon: a case study from the Paragominas region of the state of Para. *Biotropica* 21, 98–106.
- Van Rheenen, H.M.P.J.B., 2005. The role of seed trees and seedling regeneration for maintenance in logged over forests: a study in the Bolivian Amazon rainforest. Dissertation. Utrecht University, The Netherlands.
- Zuidema, P.A., Boot, R.G.A., 2002. Demography of the Brazil nut tree (*Bertholletia excelsa*) in the Bolivian Amazon: impact of seed extraction on recruitment and population dynamics. *Journal of Tropical Ecology* 18, 1–31.