

THE ECOLOGICAL AND ECONOMIC VIABILITY OF SMALLHOLDER TIMBER
MANAGEMENT IN THE AMAZON ESTUARY

By

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A DISSERTATION PRESENTED TO THE GRADUATE SCHOOL
OF THE UNIVERSITY OF FLORIDA IN PARTIAL FULFILLMENT
OF THE REQUIREMENTS FOR THE DEGREE OF
DOCTOR OF PHILOSOPHY

UNIVERSITY OF FLORIDA

2010

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Escapamos a fome da onça, fugimos dos bufalos selvagens, sobrevivemos a picada da cobra, o corte da serra afiada, aguentamos a fome em lugares distantes, muito carapanã, agua e lama, e sofremos com a distancia dos que amamos.

To my Guanabana, Dona Rosaria, Dona Raimunda, my parents, and the Mazagão River



ACKNOWLEDGMENTS

First and foremost, I would like to thank my wife for her encouragement, support and in general making life during dissertation writing as pleasant as could be. Thanks to all people of the Mazagão River who received me with open arms and taught me so much beyond their wisdom of the forest- Itamar, Geroncio, Nonato, Tomé and family, Dona Zuleide, Seu Martel, Rubilota, Seu Amilton, Jurací and many, many others. Um agradecimento especial para Dona Rosaria, Dona Raimunda e sua familia, o meu lar e familia na Amazônia hoje e sempre. Thanks to the time and valuable feedback of all my committee members, especially to my advisor Daniel Zarin. Thanks to all my Gainesville friends, many of whom have come and gone, that made my time in Florida a truly wonderful and memorable part of my life- naming names would unfortunately risk me exceeding dissertation size limitations, but you know who you are. Lastly, thanks to Monkey for her trusty company during most of my writing.

This research would not have been possible without the generous support from a Tropical Conservation and Development (TCD) fellowship and research grant, a NSF IGERT Working Forest in the Tropics fellowship and research grant, a NSF doctoral dissertation research improvement grant, a NSF South East Alliance for Graduate Education and the Professoriate (IGERT) fellowship and an EPA Science to Achieve Results (STAR) fellowship. Thank you for believing my research was possible and that I was not entirely crazy.

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Abstract of Dissertation Presented to the Graduate School
of the University of Florida in Partial Fulfillment of the
Requirements for the Degree of Doctor of Philosophy

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December 2010

Chair: Daniel J. Zarin

Major: Forest Resources and Conservation

Amazonian tidal floodplain forests were among the first to be exploited by Europeans and represent the oldest commercial logging frontier in the region. Today, thousands of families rely on timber from the tidal floodplain forests as a source of income. They harvest timber manually, without legal sanction, and process it locally at any of the hundreds of smallholder-owned micro sawmills in the area. Some previous researchers suggest that smallholders are producing timber sustainably in the region, while others have argued that timber resources are being depleted. However, past assessments lack the quantitative analyses needed to evaluate these claims critically.

This dissertation tackles the issue of the sustainability of smallholder timber use from multiple perspectives and scales. In the Mazagão watershed of the Amazon Estuary, I analyzed and modeled patterns of tree community composition and timber species population dynamics (Chapters 2 and 3); used simulation modeling to define sustainable timber management (STM) regimes for local stands (Chapter 4); and used financial return models to explore the economic viability of micro firms (Chapter 5).

Lastly, I integrated previous ecological, management, and economic models (Chapters 3, 4, and 5) to create an interdisciplinary watershed model (Chapter 6) to evaluate the future of local forests and the local timber industry under alternate scenarios of management, legality, and production levels.

While the majority of STM-related research in the Amazon has revealed discouraging prospects for both sustainable forest management and STM, results from this research show that, with changes in management and adequate institutional support, a sustainable model for smallholder timber use in the estuary is possible. Unfortunately, over centuries of mismanagement, the long history of timber extraction in the Amazon Estuary hides a gradual process of resource depletion as preferred species are sequentially exhausted. If such practices do not unchanged, the prospects for long-term management will decrease as the densities of preferred high-value species (e.g., *Carapa guianensis*, *Virola surinamensis*) fall to critically low levels that make management economically unattractive at smallholder scales.

This research demonstrates that conceptually simple models can help reveal management options and explore their projected consequences at watershed and landscape scales. Such models can lead to management that yields better ecological and economic outcomes that consider regional-specific tree community composition and ecology.

CHAPTER 1 INTRODUCTION

Despite decades at the center of global environmental concerns, the crisis of tropical forest loss endures (Hansen et al. 2010). The net effect has been massive biodiversity loss, dubious gains in local welfare and development, and massive carbon emissions that equal that of the entire global transportation sector (IPCC 2007, Hubbell et al. 2008, Rodrigues et al. 2009).

Years of divisive debates regarding the appropriate strategy to stem the ongoing crisis have focused on false choices between preservation and conservation. On one side, some argue strict preservation would leave large forest areas at risk of conversion and ignore the needs of forest inhabitants (Schwartzman et al. 2000). On the other, researchers have expressed doubt about production-based forest conservation strategies (Homma 1992, Redford and Sanderson 2000, Rice et al. 2001). However, it is increasingly clear that the ecological and social complexity and variability of tropical forests require a flexible approach where preservation and conservation are coupled with a more comprehensive valuation of forest goods and services.

Studies of forest succession, along with evidence of pre-Colombian large-scale forest conversion in seemingly intact forests, indicate that tropical forests are more ecologically resilient than generally acknowledged (Uhl et al. 1988, Willis et al. 2004, Heckenberger et al. 2007). Forests seem able to recover structurally and compositionally (in terms of richness) within years-to-decades from small-scale human disturbances such as logging gaps and small agricultural plots. To acknowledge the long history of human disturbance and powerful regenerative ability of tropical forests does not mean humans should exploit forests without concern. On the contrary, it

shows we cannot take a snapshot of a forest that holds no physical sign of human interference today and hold it as the single measuring stick. It is to recognize tropical rain forests as dynamic entities and people as shapers of past and present environmental change.

From a conservation perspective, sustainable forest management (SFM) at its core attempts to curb deforestation by making low-impact forest-based activities competitive over other destructive land uses. However, SFM has been an illusive goal as its competitiveness depends on a set of ecological, economic, and social conditions that must be maintained indefinitely to ensure forest persistence (ITTO 2005, UNFF 2009).

The Potential Role of STM in SFM

According to the ITTO (2006), only 7% of areas under timber use are managed sustainably. Yet, across most tropical forests, the viability of long-term timber management is still unexplored. For remote tropical areas where timber is a main product, this means sustainable timber management (STM) that ensures yields harvest after harvest should be an important component of SFM (Seydack 1995; Zarin et al. 2007 but see Luckert and Williamson 2005). With 25% of world's poor being forest dependent and an estimated 0.5-1 billion smallholders managing trees and remnant forests, the need for SFM and STM at community and smallholder scales is clear (Scherr et al. 2004). Unfortunately, most forest legislation focuses on administrative requirements, fees, taxes and property rights that pose challenges to small holders and are not directly related to SFM (Kaimowitz 2003), and most tropical forest management research has focused on large-scale industrial operations.

In the Amazon Estuary, smallholders have developed a micro-scale vertically-integrated system of timber production (Pinedo-Vasquez et al. 2001, Sears et al. 2007a). Contrary to community-led efforts elsewhere, these informal micro firms are owned individually and commonly integrate timber extraction and processing in local circular sawmills. For decades, hundreds of these micro firms have produced sawn lumber sold primarily at local and regional markets (Barros and Uhl 1995). As an example of a small-scale localized timber production system, the Amazon Estuary is ideal for exploring issues of smallholder vertically-integrated timber use.

Timber, Smallholders and the Amazon Estuary

Floodplain forests cover 25,000 km² of the Amazon estuary (Lima, 1956). Compared to the terra firme forests, floodplain forests have a lower diversity of tree species and a high abundance of economically valuable species (Lopez & Kursar, 2003; Martin et al., 1992; Parolin et al., 2004; Terborgh & Andresen, 1998). Low floodplain forests may experience tidal flooding from fresh water back up twice a day, while higher areas may experience flooding only in the wettest months between January through May. After centuries of disturbance, the persistence of Amazon tidal floodplain forests indicates a potential for sustainable timber management (Raffles, 1997). This potential is further bolstered by an abundance of timber, relatively fertile soil, and low cost/ low damage of water transport (Anderson 1990, Zarin et al. 1998). In relation to timber harvesting in Amazonian upland forests, floodplain forest operations have lower negative environmental impacts due to the lack of heavy machinery use and the reliance on river transportation that precludes road construction and eventual deforestation (Kaimowitz and Angelsen 1998, Laurance 2001). Additionally, the dangerous fire-feedback mechanisms that often hinder the recovery of upland forests

following logging are absent in floodplain forests (Nepstad et al. 1999, Gerwing 2002, Cochrane and Laurance 2002).

Caboclos are the traditional inhabitants with mixed indigenous, European and African descent and show a detailed knowledge of their environment. *Caboclos* have developed ecologically-integrated livelihood strategies through centuries of coexistence with the Amazonian floodplains (Hiraoka, 1992). Much of the Amazon tidal floodplain has been historically inhabited by communities of caboclo smallholders. At present, timber remains an important resource for subsistence and an important source of income for many *caboclo* families. Hundreds of small-scale, family-run timber operations supplied by many more thousands of smallholders with forest resources rely on timber as a source of income (Barros & Uhl, 1995; 2005).

Past research on *caboclo* small-scale timber production has documented cases of apparently sustainable forest use in the region (Pinedo-Vasquez et al. 2001, Sears and Pinedo-Vasquez 2004), while others have argued that timber resources are being depleted (Macedo and Anderson 1993, Barros and Uhl 1995). But these assessments lack quantitative analyses needed for accurate evaluation of impacts of past, present, and future forest use.

Scope of Dissertation

In this dissertation I evaluate the sustainability of timber production in the region based on modeling watershed-scale timber resource availability and use. A watershed approach allowed me to incorporate the current extent of forest cover types compatible with timber production within the watershed, the ecology and yield of species utilized, and the economics of timber production. Results are useful for the evaluation of current

and future scenarios of timber extraction intensity and methods while providing insights into the main factors limiting sustainable timber use in the Amazon estuary.

My research is primarily focused in the Mazagão watershed at the Western side of the Amazon estuary (Figure 1; Chapters 3-6). This 160 sq km watershed has a long history of timber use and is similar in composition and land use history to several adjacent watersheds. Preliminary surveys in the study area show timber as the base of a complex economy where some households will often extract and process timber in family-run sawmills from the land of other households within the watershed. Because of this, the study of timber use and management at stand and property scales is inappropriate since timber-related activities commonly happen across property boundaries and over multiple forest stands. A watershed scale approach is also more appropriate because the *caboclo* community that extracts, processes, and markets timber resources of the watershed is entirely enclosed within the watershed, facilitating linking ecological and social processes at a compatible scale.

Component 1: From Regional to Watershed

Most forest use in the Amazon estuary now occurs on plots of < 100 ha (Sears and Pinedo-Vasquez 2004); region-wide generalizations are not particularly relevant to this scale unless conditions are homogeneous.

In Chapter 2, I examine stands selected to represent what is characterized as intact forest by local people, whose primary use of such areas is for hunting and limited harvesting of fruit from unmanaged açai palms. Results illustrate high, estuary-wide variability in forest structure and composition among apparently similar estuarine floodplain forests, and emphasize how local structural and compositional differences

result in contrasting potential for sustainable forest management. This chapter serves as a regional context for the other chapters.

Component 2: Timber Species Ecology, Population Dynamics, and Long-Term Timber Use in an Amazon Tidal Floodplain Watershed.

The absence of tree demography data has inhibited the formulation and evaluation of tidal floodplain-specific guidelines for ecologically sound forest management. In Chapter 3 I evaluate species-specific abundance, diameter and spatial distributions, and recruitment, growth and mortality rates of timber species in tidal floodplain forests. I also employ matrix population models to explore the underlying population dynamics of common tidal floodplain species and explore the link between species population ecology and past, present, and future timber use. In my analysis, I combine conventional tree demography methods with data from interviews with tidal floodplain *Caboclos* regarding land use and ecological knowledge.

While a wide choice of models have been applied to address the ecological viability of STM (Vanclay 1995, Kammesheidt et al. 2001, Glauner et al. 2003, Phillips et al. 2004, Gourlet-Fleury et al. 2005, Valle et al. 2007), most relevant research has focused on the evaluation of harvest damage, post-harvest effects on growth, recruitment and mortality separately (Fredericksen and Mostacedo 2000, Chapman and Chapman 1997, Finegan and Camacho 1999). Without integrating these population demography responses, however, net population effects of timber harvest and management remain unclear. Furthermore, most related research to date has focused on evaluating current practices instead of searching for alternate sustainable management regimes relevant to forest managers and regulators.

In Chapter 4 I used a matrix-based harvest simulation model to evaluate the prospects for sustained timber production in the Amazon estuary. Based on a wide range of values for management criteria (e.g., length of cutting cycle, minimum cut diameter), I projected the population and yield outcomes of hundreds of long-term timber management regimes. These results were then compared using simple quantitative indicators of STM to find optimal stand-level and species-specific sustainable timber management regimes relevant to similar forests in the Amazon Estuary.

Component 3: Timber Resource Use and Economics in an Amazon Tidal Floodplain Watershed.

In order to understand the potential for timber management, we must not only consider the ecological limitations to management but also the economic viability of timber harvesting (Boot, 1997; Buongiorno et al., 1995; Lu & Buongiorno, 1993). Smallholder timber operations may vary substantially from industrial operations in techniques, technology, capital availability, market reach and ecological impacts (chapter 4, Salafsky et al. 1998, Rockwell et al. 2007a, Keefe 2008). Previous research has estimated that there are nearly 800 family-run sawmills in the Amazon Estuary that function under a different set of costs and limitations than the larger operations commonly found in other areas in the Amazon (Barros & Uhl, 1995; 2005; 2003). In the Mazagão watershed, 12 timber micro firms were responsible for processing the harvested wood from the watershed. Surprisingly, relatively little attention has been paid to the potentials and limitations of timber management at smallholder scales, with the most relevant research focusing on community forestry efforts (d'Oliveira 2000, Rockwell et al. 2007b). Without the proper knowledge and consideration of the

potentials and limitations of smallholder timber management, most regulation of timber use in the tropics has focused on the industrial scale, leading to unrealistic expectations for smallholders and communities (d'Oliveira 2000, Rockwell et al. 2007a, Zarin et al. 2007).

In Chapter 5 I use data from multiple sources including landowner and firm surveys, participatory monitoring of firms, and detailed forest and sawmill operation monitoring to devise a financial returns model of smallholder timber micro firms and a simpler model for smallholder açai fruit production. I then explore the economics of timber micro firms to address the following questions:

- (1) What are the financial costs and revenues of timber micro firms?
- (2) What micro economic factors most influence long-term economic viability of timber production by micro firms?
- (3) How does timber micro firm profitability compare to regional economic alternatives? How can this be used to advance tropical conservation and development?

Component 4: Integrating Ecological, Management and Economic Models at the Watershed Scale to Explore the Prospects of Management and Long-Term Forest Use.

With a few exceptions, the ecologic and economic viability of production in tropical forest has been evaluated at single firm/ single stand scales. However, to effectively balance conservation and economic viability necessary for sustainable tropical forest use, ecological and economic limitations must be considered simultaneously (Boot and Gullison 1995, Merry et al. 2009).

In Chapter 6, following the previous chapters exploring community composition (Chapter 2), timber species population ecology (Chapter 3), sustainable timber management (STM) regimes for local stands (Chapter 4) and the economic viability of

micro firms (Chapter 5), I create an interdisciplinary model to evaluate watershed timber resource availability and use under alternative scenarios of management, legality, and production levels. We employed multiple forest use scenarios to address questions of sustainability at the scale of a whole watershed (and its related local timber industry) to explore the link between ecological, economic and legal constraints to forest use and proposed management.

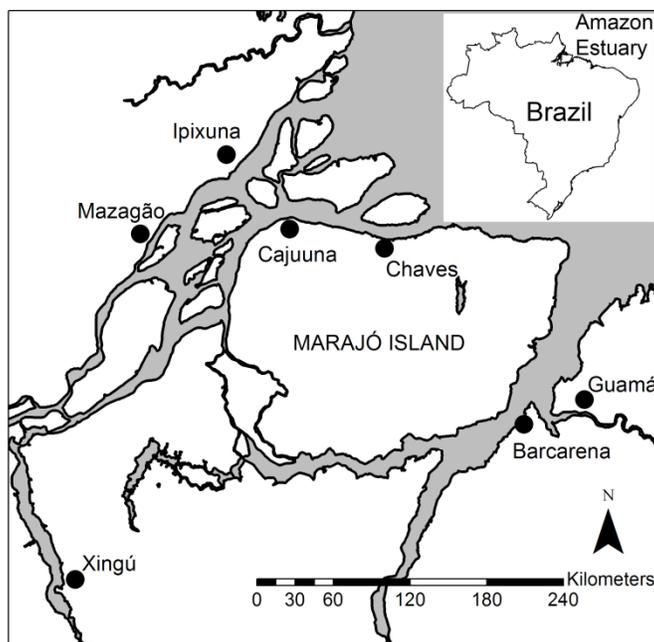


Figure 1-1. Map of Amazon Estuary with location of areas contemplated in the dissertation. The regions of Ipixuna, Guamá, Barcarena, Cajuuna, Chaves and Xingú are considered only in Chapter 2. All other chapters focus on Mazagão watershed as a comprehensive ‘microcosm’ study.

CHAPTER 2 MIXED POTENTIAL FOR SUSTAINABLE FOREST USE IN THE TIDAL FLOODPLAIN OF THE AMAZON RIVER

Background

The tidal floodplain forests of the Amazon estuary have supported timber harvesting since at least as early as the 17th century, but in a much more selective form (in size and species) than modern extraction. Beginning in the late 1950s, large saw and plywood mills were established in the region during a period of intense exploitation that lasted until the 1980's. During this period *Cedrela odorata*, *Virola surinamensis*, *Carapa guianensis*, *Hura crepitans*, and *Maquira coriacea* were the most sought-after species (Raffles 1999). Since then, small-scale logging and milling operations have replaced the larger sawmills (Barros and Uhl 1995, Pinedo-Vasquez et al. 2001).

The persistence of logging activity in the region suggests an underlying potential for sustainable management of timber production, potential that is bolstered by a high number of timber species, the relatively low impact of extraction due to water transport, relatively fertile soils, and well-developed local ecological knowledge (Anderson 1990, Hiraoka 1992, Zarin et al. 1998, Laurance 2001). While some researchers suggest that smallholders are managing timber production sustainably in the region (Pinedo-Vasquez et al. 2001, Sears and Pinedo-Vasquez 2004), others have argued that sustainable management is largely unrealized and that timber resources are being depleted (Macedo and Anderson 1993). Additionally, non-timber forest products, especially the açai palm (*Euterpe oleracea*), have become increasingly important in the subsistence and market economies of floodplain inhabitants (e.g., Brondizio 2004). It is

unclear whether non-timber forest products are complementary or incompatible with timber production and forest conservation (Alavalapati and Zarin 2004).

Most forest management in the Amazon estuary now occurs on plots of < 100 ha (Sears and Pinedo-Vasquez 2004); region-wide generalizations are not particularly relevant to this scale unless conditions are homogeneous. Within the Amazon estuary, there are few cross-site comparisons (de Almeida et al. 2004). Previous regional analyses of vegetation cover in the estuary were unable to detect subtle compositional differences with remote sensing (Zarin et al. 2001, Pereira et al. 2002), or lack consideration of natural variability of forest composition at the local property or community scale relevant to riverine forest management (Brondizio et al. 1994). Most previous research on tidal floodplain forest has focused on the description and viability of currently managed forests (Anderson 1986, 1991, Anderson et al. 1995, Munizmiret et al. 1996, Brondizio and Siqueira 1997, Sears and Pinedo-Vasquez 2004). In this study, we examine stands selected to represent what is characterized as intact forest by local people, whose primary use of such areas is for hunting and limited harvesting of fruit from unmanaged açai palms. We illustrate the degree of heterogeneity among apparently similar estuarine floodplain forests, and emphasize how local structural and compositional differences result in contrasting potential for sustainable forest management.

Study Area

We conducted our research at two tidal floodplain forest sites in the Amazon Estuary near Macapá, the state capital of Amapá. Mean annual temperature is 27C and average daily temperature varies by less than 3C from month to month in the region. Mean annual precipitation is 2550 mm and occurs mostly in the wet season months of

January-May. This part of the Amazon estuary is characterized by freshwater tidal fluctuations of 2-3 m. The two sites, Mazagão (00 03' N and 50 37'W) and Ipixuna (00 30' S and 51 13'W), are southwest and northeast from Macapá, respectively; both sites are on tributaries near the main stem of the Amazon River's north channel (Figure 2-1). Because of the elevated river level in the wet season, both sites are flooded twice daily during high tide.

We also used published data from two other studies to compare our results within the larger context of intact Amazon estuarine forests. Data from an early 3.8 ha inventory of a tidal floodplain forest by the Guamá River (Pires and Koury 1959) was included to compare the composition of Mazagão and Ipixuna plots with that of an old growth forest site. Although Pires and Koury did not include formal collection of land use history for their inventory area, they characterize the stand as primary forest. In addition, we included four 1-ha estuary forest plots from Almeida et al. (2004) to expand our geographical extent of compositional comparisons. These plots were located across the Amazon Estuary in the municipalities of Barcarena, Afuá (Cajuuna Island), Chaves, and at the lower Xingú River.

Methods

At each research site, five 1 ha plots were located within a 75 ha area in Ipixuna and 90 ha area in Mazagão. Plots were not located randomly at each site due to limited available areas for permanent plot establishment. Instead, plots were selected to represent intact forest. Diameter measurements and species identifications were made for all stems > 5 cm DBH within each plot (hereafter "overstory"). Stems > 10 cm height and < 5 cm DBH were identified and counted within 5 X 5 m subplots (n = 16 per plot, hereafter "understory"). The Guamá inventory included all trees with DBH > 10 cm in a

380 x 100 m strip (Pires and Koury 1959). For the purposes of comparing Guamá composition with that of Ipixuna and Mazagão on an equal area basis, we divided the Guamá data into three 1 ha separated by 40 m buffers. The four 1 ha plots sampled by Almedia et al. (2004) also included only trees with DBH > 10 cm. Interviews with landowners, loggers, and sawmill operators were carried out from May to August, 2004 in Mazagão and Ipixuna. Questions regarding the historical use of local forests and the distribution of commercial species were asked to evaluate land use history and local perception of patterns of forest composition within and between Mazagão and Ipixuna.

To compare forest management potential between areas, we separated inventoried species into palm and non-palm (eudicots) categories and separated timber species as a separate sub-category within the non-palm group. Density, frequency, dominance, species richness, Sorensen distance, Simpson, Shanon and evenness indices and measures were calculated using standard methods and formulas (Nebel et al. 2001, McCune and Grace 2002) . Species accumulation curves, ordinations, indicator species analysis, multi-response permutation procedure (MRPP), Simpson diversity and evenness and Sorensen distance were calculated using PC-ord software (version 4, SAS institute inc.) following procedures suggested by McCune and Grace (2002). Species accumulation curves were created for each region by classifying individuals measured as unique data entries and then calculating average richness for all subsamples from $n=1$ until $n=N$, with 500 randomly picked replicates for each sample size.

We used detrended correspondence analysis (DCA) of the log transformed abundance data [$\log(x + 1)$] for ordination purposes because it produced the most

stable and clearly interpretable results. The use of DCA in our ordination analyses allowed us to look at how species are positioned within ordination space. We hypothesized that, if forest composition were influenced by differences in land use history, species ordination along the main axis that separated plots from the two areas would follow a gradient of successional status. We used the same technique to test if other life traits, such as light and soil moisture requirements influenced the position of species along axes of our DCA solution. We acquired life trait information from field observations, interviews with local forest users about species distributions, and from the available literature about the species (Vera et al. 1999, Webb 1999, Lopez and Kursar 1999, Adler and Kielinski 2000, Lorenzi 2000, Sears 2003).

We also inspected the clustering of species in ordination space and performed cluster analysis to identify possible species associations and related life history trait similarities. No species or plots were picked out by multivariate outlier analyses (Pc-ord version 4, SAS institute inc.) and hence all data were included in the analysis. All comparisons between Mazagão and Ipixuna were done at the species level. Biomass was calculated for all 1 ha plots in Mazagão and Ipixuna using DBH-whole tree biomass regression equations from Johnson (1999) to all overstory stems. These equations were derived from a tidal floodplain forest site at Mazagão.

$$\text{Palm biomass} = 0.0052 \times \text{DBH}^{3.74} \quad (R = 0.993, n = 6)$$

$$\text{Virola biomass} = 0.0481 \times \text{DBH}^{2.55} \quad (R = 0.985, n = 5)$$

$$\text{Dicot (no Virola) biomass} = 0.1155 \times \text{DBH}^{2.501} \quad (R = 0.985, n = 18)$$

Given the very high relative frequency of palms in all Mazagão and Ipixuna plots (accounting for 47 and 72% of individuals >5cm DBH, respectively), and their commercial importance, we looked at the possible relations between palm abundance and overall forest structure and composition. We looked specifically at the relationship between the two most abundant palms in the study, *A. murumuru* and *E. oleracea*, and forest structure and composition since they accounted for almost all palm individuals measured in the study.

We performed all comparisons between Mazagão, Ipixuna, Guamá and the Almeida et al. plots at the genus level which makes compositional similarities between sites more apparent (Terborgh and Andresen 1998). Genus level comparisons were used because of the difficulty in tracking changes in names for the species present in the Guamá inventory and because of the geographic distance between the areas. Additionally, only individuals with DBH > 10 cm were used in this analysis because of the differences in minimum DBH between our methods and those of the other two studies.

Results

Total stem density > 5 cm dbh was 15% lower for Mazagão than for Ipixuna, but 35% higher for Mazagão for non-palm species. A similar pattern is evident in the understory data- 20% lower total stem density for Mazagão, but 50% higher for Mazagão when palm species were excluded (Table 2-1). These differences result from larger populations of *A. murumuru* and açai palms at Ipixuna than at Mazagão (Appendix). Stem density of commercial timber species was more than twice as high at Mazagão than at Ipixuna, both for all stems > 5 cm dbh and for large timber individuals (> 30 cm dbh); density of commercial timber species < 5 cm was three times greater at

Mazagão. Mean basal area and biomass values were higher in Mazagão, but only the basal area differences were statistically significant.

A few timber species showed clear differences in their size class distribution between plots from Mazagão and Ipixuna (Figure 2-2). For instance, while *C. spruceanum* size distribution showed an inverse-J shape in Ipixuna, the distribution of the same species in Mazagão was characterized by much fewer individuals in the smaller size classes but many more individuals in the larger size classes. On the other hand, this pattern was inverted for *Licania heteromorpha*, a species currently not utilized in the either site. Additionally, *Carapa guianensis*, *Mora paraensis*, *Pentaclethra macroloba* and *Quararibea guianensis* were practically absent from all Ipixuna plots but present in all Mazagão plots with individuals as large as 60cm DBH.

Mazagão plots had a combined total of 88 overstory species, with a mean of 51 ha⁻¹, while Ipixuna plots contained fewer species, having a combined total of 67 overstory species, and a mean of 36 ha⁻¹. For understory individuals Mazagão had a cumulative species richness of 68 and Ipixuna had 42. Simpson's diversity and evenness values were higher in Mazagão for both strata, but only the overstory differences were statistically significant (Table 2-2). Species-accumulation curves also reflect a higher diversity of Mazagão plots (Figure 2-3).

The percentage of tree species represented by only one individual in each area was similar between Mazagão and Ipixuna (23% v 24%, respectively). 31% of trees species in Mazagão were represented by only 1-2 individuals, compared to 40% of Ipixuna species. 41% of understory species in Mazagão were represented only by 1-2

individuals while only 26% of species in Ipixuna's understory were represented only by 1-2 individuals.

The two areas shared 5 of the 10 most abundant tree species (*E. oleraceae*, *A. murumuru*, *C. spruceanum*, *L. heteromorpha*, and *T. surinamensis*), with *E. oleracea* and *A. murumuru* being the first and second most abundant species in both areas. Four of Mazagão's 10 most abundant species were commercial timber species (*C. spruceanum*, *L. heteromorpha*, *P. maculosa*, and *G. augusta*), whereas the 10 most abundant species from Ipixuna included only two commercially harvestable species (*C. spruceanum* and *L. heteromorpha*). Species list and abundance for Mazagão and Ipixuna sample plots are provided in appendix A.

Indicator species analysis yielded 21 tree species that are significantly associated to Mazagão and 9 species that were associated to Ipixuna (Table 2-3). Nine of Mazagão's indicator species were of commercial logging value while none of Ipixuna's indicator species were harvestable.

Separation between Mazagão and Ipixuna plots in our DCA results occurs along in the primary axis representing the strongest gradient in the data set. This separation was clearer when ordinating tree and understory layers in separate analyses (Figure 2-4). The positioning of early- to mid-successional species along the primary axes of the DCA ordination of tree data reveal no successional gradients between Mazagão to Ipixuna sites. DCA results also show there is no clear gradient in species soil moisture or light requirements along the ordination axes (Figure 2-5). The ordination of species data also revealed no isolated clusters of species in ordination space, signaling no clear species grouping besides the vertical clustering that happened on either extreme of the primary

axis which represent species that happened exclusively in either of the two areas; also, the additional cluster analysis did not reveal any clear groups of common life history traits.

Timber basal area was significantly correlated with palm basal area at the plot level ($r = -0.66$, $p < 0.05$; Figure 2-6). Non-palm basal area was strongly correlated to *A. Murumuru* basal area ($r = -0.89$, $p < 0.001$; Figure 2-6). *E. oleracea* abundance had no clear relation to forest structure and composition.

The Pearson product moment correlation between plot-level non-palm total abundance and *A. murumuru* abundance yielded a significant negative relationship ($r = -0.8625$, $p < 0.01$). Plot-level richness and diversity were also negatively related to *A. murumuru* abundance. For tree data, the abundance of *A. murumuru* was negatively related to richness ($r = -0.8414$, $p < 0.01$) and Simpson diversity ($r = -0.7240$, $p < 0.05$) at the plot level. For this correlation analysis, we excluded *E. oleracea* and *A. murumuru* data from the Simpson calculation to avoid the strong impact of palms in the results and the autocorrelation of the Simpson index to *A. murumuru* abundance. A weaker relation between palm abundance and species composition of the understory was also observed where results resemble patterns of palm impact on overstory composition. *A. murumuru* abundance had a negative correlation to understory richness ($r = -0.6260$, $p < 0.05$).

The two-dimensional DCA solution with Mazagão, Ipixuna, Guamá and the Almeida et al. data show a clear compositional separation of plots by region. These differences reflect the geographical distribution of the sampled regions (Figure 2-1 and Figure 2-7). Only five genera were present in all of the seven regions including *Virola*

(Myristicaceae), *Inga* (Mimosaceae) and *Ficus* (Moraceae). Neither the *Euterpe* or *Astrocaryum* genera, arguably the most abundant genera across the estuary, were present in all of the seven regions.

When compared to Mazagão and Ipixuna plots, the composition of the unlogged Guamá plots were more similar to the composition of Mazagão plots as shown by the two-dimensional DCA solution (Figure 2-7). The first axis of the solution clearly divides each of the three areas while the second axis explains primarily compositional variation in Guamá plots. The results from the DCA analysis were supported by the multi-response permutation procedure (MRPP) that shows a clear significant compositional separation between the 3 sites ($p < 0.00001$) and the smaller average Sorensen distance between Mazagão and Guamá plots compared to Ipixuna and Guamá plots and Mazagão and Ipixuna plots.

Discussion

Mixed Potential for Contrasting Forest Uses

Although tidal floodplain forests have been commonly perceived as a homogenous forest type (Hiraoka 1992), these forests vary greatly in forest composition and structure between individual stands and sub-regions (Martin et al. 1992, Anderson et al. 1995). For the typical estuarine smallholder or riverine community, this local variability of forest composition results in widely differing potential for forest management (Table 2-4).

The majority of the most important saw timber species used today (*Carapa guianensis*, *Callycophyllum spruceanum*, *Mora paraensis* and *Pentaclethra macroloba*) had noticeably higher abundances in Mazagão compared to Ipixuna. Most of the other saw timber species had similarly low abundances for both regions. In terms of plywood production, neither site has good potential for plywood management given the low

abundance of plywood species (e.g., *Ceiba pentandra*, *Olmedia caloneura*, *Virola surinamensis*, *Calophyllum brasiliense*, *Hura crepitans*), especially in the larger size classes. However, Mazagão has a noticeably higher potential for management for seed oil production given its high abundances of *Carapa guianensis* and *Pentaclethra macroloba*, both important oil producing species that were practically absent from Ipixuna. All of the potential for management for seed oil in Ipixuna is related to a high density of *A. murumuru* palms, but although extraction of *A. murumuru* fruits was an important activity from the earlier part of the 20th century to the 1970s, it is currently not used due to lack of outside demand. Ipixuna only has a slightly higher management potential for fruit production having a greater abundance of *Euterpe oleracea*, *Spondias mombin* and *Theobroma cacao*. Smallholders are well aware of local differences in forest composition and value land holdings according to local forest composition.

Palm Dominance and Alternative Forest Uses

One important variable in tidal floodplain forest composition is the degree of dominance of palm species. The dominance of palms in tidal floodplain varies from the monospecific palm stands to diverse forests with understory palms (Zarin et al. 2001). Our results show that sites with low palm abundance are more suitable for timber management. However, açai palm distribution is not related to observed variation in forest composition and structure, suggesting that tidal floodplain forests may be sustainably managed for both timber and açai production. This results is of special importance given that timber and açai are the two most important forest products for smallholders in the estuary, and is consistent with reports of extant sustainable dual production systems described by Pinedo-Vasquez et al. (2001).

There is concern that the high price of açai palm fruit will gradually transform the tidal floodplain forests into açai-dominated palm forests due to the preferential management of açai over other floodplain species (Brondizio et al. 1994, Brondizio and Siqueira 1997). Several studies show how the management for açai fruit production greatly increases the dominance of the palm often at the detriment to other species including timber (Anderson 1986, 1990, Brondizio 2004). In sites like Ipixuna where there are few other attractive forest-based land uses, land holders are increasingly managing forests for açai production. In fact, land cover change analysis show an increase in palm forest area in Ipixuna between 1970s to 1990s (Zarin et al. 2001). Although palm forest areas have expanded in and around Mazagão during the same period, today the diversity of livelihood strategies in Mazagão reflect the potential for multiple forest uses including many family-run sawmills. The relatively high potential for timber management in Mazagão also is reflected in recent descriptions of sustainable management of tidal floodplain forests of Mazagão, despite the past intensive logging boom that strongly impacted local forests from 1950-1980s (Pinedo-Vasquez et al. 2001, Pinedo-Vasquez and Rabelo 2002, Sears and Pinedo-Vasquez 2004). Mazagão has a recent logging history that dates back to the beginning of the 19th century when steam ships would purchase high-valued species for export, through the logging boom of the 1950-1990s, through the current thriving smallholder-dominated industry in Mazagão. Ipixuna on the other hand had its modern period of timber extraction lasting from the opening of a small-sized sawmill in the region in the 1950s until its closing in the 1980s and since then has increasingly focused on açai production.

Concluding remarks on Amazon Estuary composition variability

Our results indicate that contrasting views of the management potential of Amazonian tidal floodplain forests may reflect not only heterogeneity of forest use by riverine peoples, but also the heterogeneity of the forests themselves. Indeed, the notion of a homogeneous tidal floodplain forest type is incompatible with the diversity of estuarine livelihood systems. The forest composition of sites similar to Mazagão facilitates sustainable management for timber and other forest uses, whereas the current management potential for timber of areas similar to Ipixuna is low. Poor understanding of potential growth and yield of tidal floodplain species prevents us from understanding the current constraints of forest management and limits our ability to devise viable guidelines and policy interventions (Sears and Pinedo-Vasquez 2004).

Table 2-1. Summary of structural differences between Mazagão and Ipixuna (Means per ha and SE)

	All trees	Non-palm species	Timber species	Large timber individuals ^a
Overstory				
Density (ind/ ha)				
Mazagão	1033 (94)	544 (35)*	285 (17)***	104 (8)***
Ipixuna	1177 (206)	329 (69)*	116 (26)***	51 (2)***
Basal area (m²/ha)				
		33.7		
Mazagão	38.3 (1.3)*	(1.4)**	24.1 (1.6)**	20.4 (1.4)*
		25.8		
Ipixuna	33.9 (1.5)*	(1.4)**	17.6 (1.1)**	16.2 (1.1)*
Biomass (Mg/ha)				
Mazagão	386 (12.9)	348 (16.7)	254 (17.6)	230 (16.4)
Ipixuna	352 (13.7)	303 (16.3)	214 (17.3)	206 (18.1)
Understory				
Density (1000 ind/ ha)				
Mazagão	24.8 (6.2)	17.5 (4.0)	7.4 (1.2)**	
Ipixuna	29.3 (4.0)	11.9 (3.0)	2.4 (0.3)**	

^a All commercial trees >30cm DBH. * $p < 0.05$, ** $p < 0.01$, *** $p < 0.001$

Table 2-2. Summary of compositional differences between Mazagão and Ipixuna (Means per ha and SE)

	Overstory	Understory ^a
Richness		
Mazagão	51.4 (2.1)**	30.8 (3.1)*
Ipixuna	36.4 (3)**	21 (1.8)*
Simpson		
Mazagão	2.59 (0.12)***	2.31 (0.25)
Ipixuna	1.77 (0.05)***	1.76 (0.16)
Evenness		
Mazagão	0.66 (0.03)**	0.67 (0.06)
Ipixuna	0.50 (0.01)**	0.58 (0.06)

^a Based on a 0.04 ha sampled area for each hectare plot. * $p < 0.05$, ** $p < 0.01$, *** $p < 0.001$

Table 2-3. Indicator species analysis for Mazagão and Ipixuna overstory data

Mazagão			Ipixuna		
Species	Indicator value	<i>P</i>	Species	Indicator value	<i>p</i>
<i>Combretum cacoucia</i>	100	0.004	<i>Chrysophyllum argenteum</i>	100	0.004
<i>Hevea brasiliensis</i>	100	0.004	<i>Chrysophyllum excelsum</i>	100	0.004
<i>Metrodorea flavida</i>	100	0.004	<i>Parinari excelsa</i>	100	0.004
<i>Mora paraensis</i>	100	0.004	<i>Calyptranthes speciosa</i>	95.4	0.01
<i>Pentaclethra macroloba</i>	100	0.004	<i>Spondias mombin</i>	94.4	0.014
<i>Pouteria sagotiana</i>	100	0.004	<i>Theobroma cacao</i>	92.6	0.004
<i>Pterocarpus amazonicus</i>	100	0.004	<i>Herrania mariae</i>	85	0.019
<i>Pterocarpus officinalis</i>	100	0.004	<i>Inga lentiscifolia</i>	80	0.049
<i>Quararibea guianensis</i>	100	0.004	<i>Astrocaryum murumuru</i>	78.2	0.004
<i>Sarcaulus brasiliensis</i>	100	0.004			
<i>Swartzia cardiosperma</i>	99.3	0.004			
<i>Carapa guianensis</i>	99.1	0.004			
<i>Pachira aquatica</i>	94.4	0.015			
<i>Tachigali paniculata</i>	93.3	0.02			
<i>Alibertia sp.</i>	80	0.049			
<i>Attalea phalerata</i>	80	0.041			
<i>Dendrobangia boliviana</i>	80	0.049			
<i>Matisia paraensis</i>	80	0.041			
<i>Pouteria bilocularis</i>	80	0.041			
<i>Zygia jruana</i>	80	0.049			

Table 2-4. Potential of alternate forest uses between Mazagão and Ipixuna, Amapá, Brazil

	Mazagão	Ipixuna
Timber		
Sawnwood	High	Low
Plywood	Low	Low
Non-timber		
Latex	Medium	Low
Seed oils	High	High ^a
Palms and other fruits	High	High

^a based on the high abundance of *A. murumuru* palm which is currently not utilized for oil production

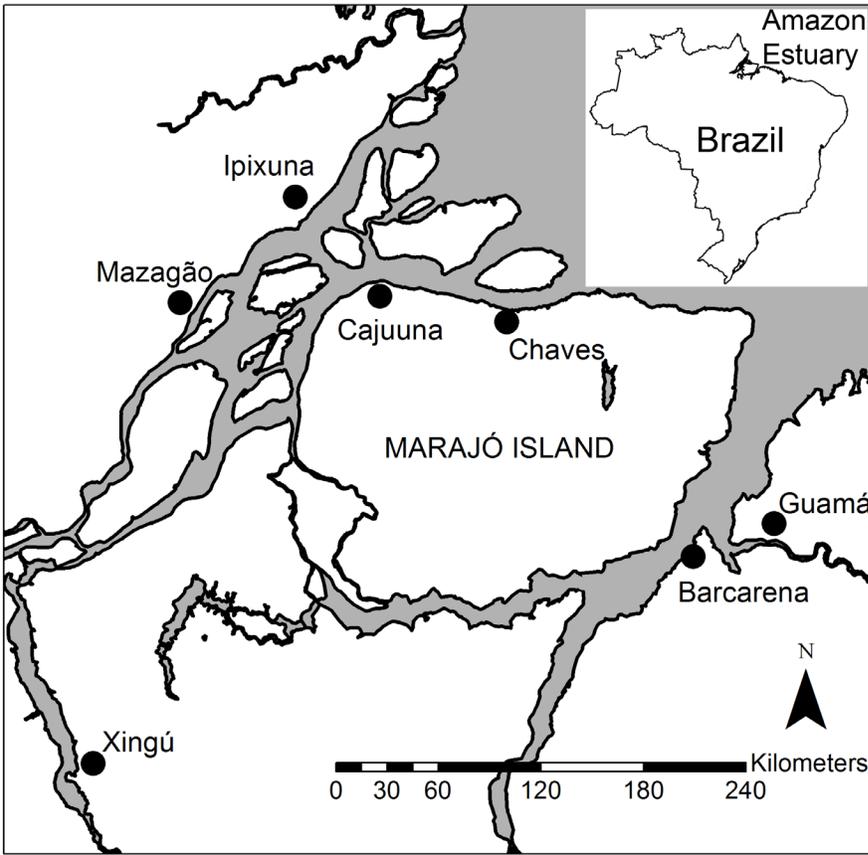


Figure 2-1. Map of Amazon Estuary with location of forest inventory plots from current study (Mazagão and Ipixuna), Pires and Koury (1959; Guamá), and Almeida et al., (2004; Barcarena, Cajuuna, Chaves and Xingú).

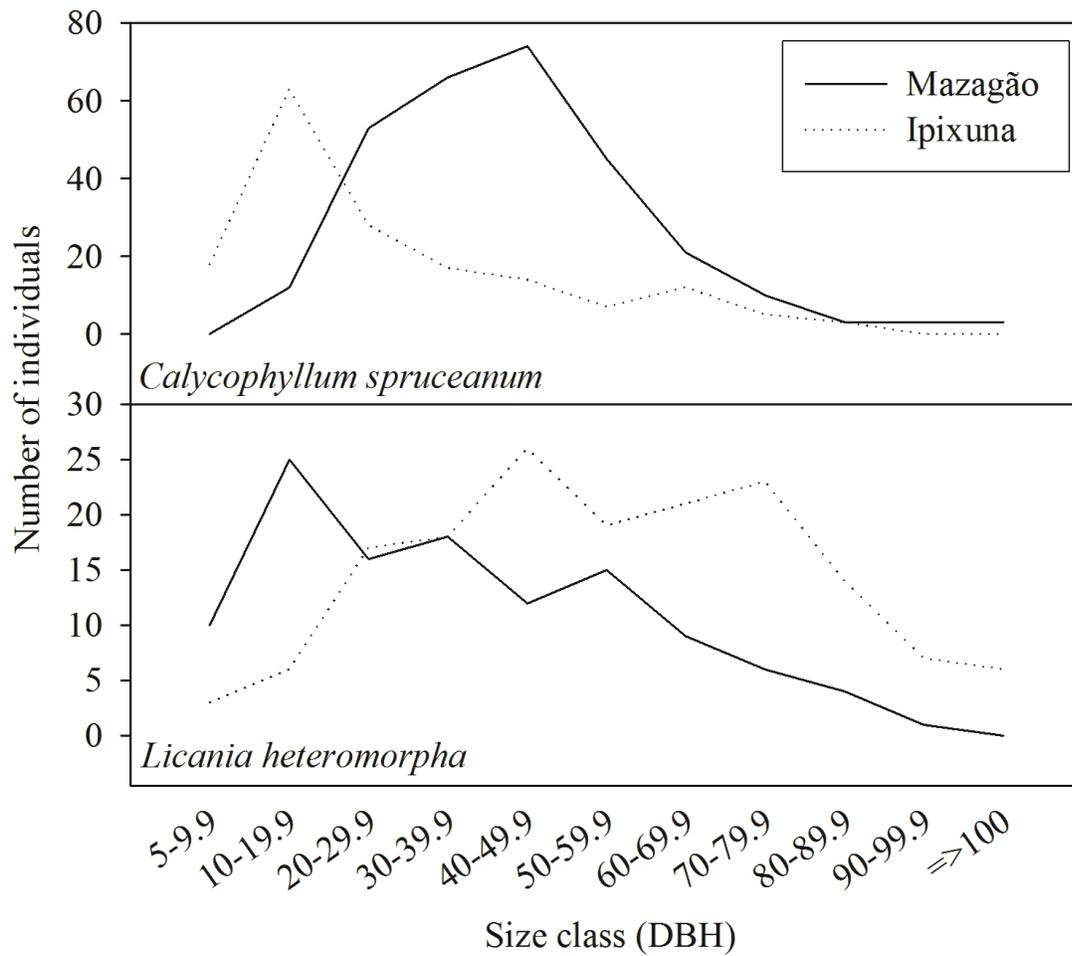


Figure 2-2. DBH distribution for harvestable species in Mazagão and Ipixuna.

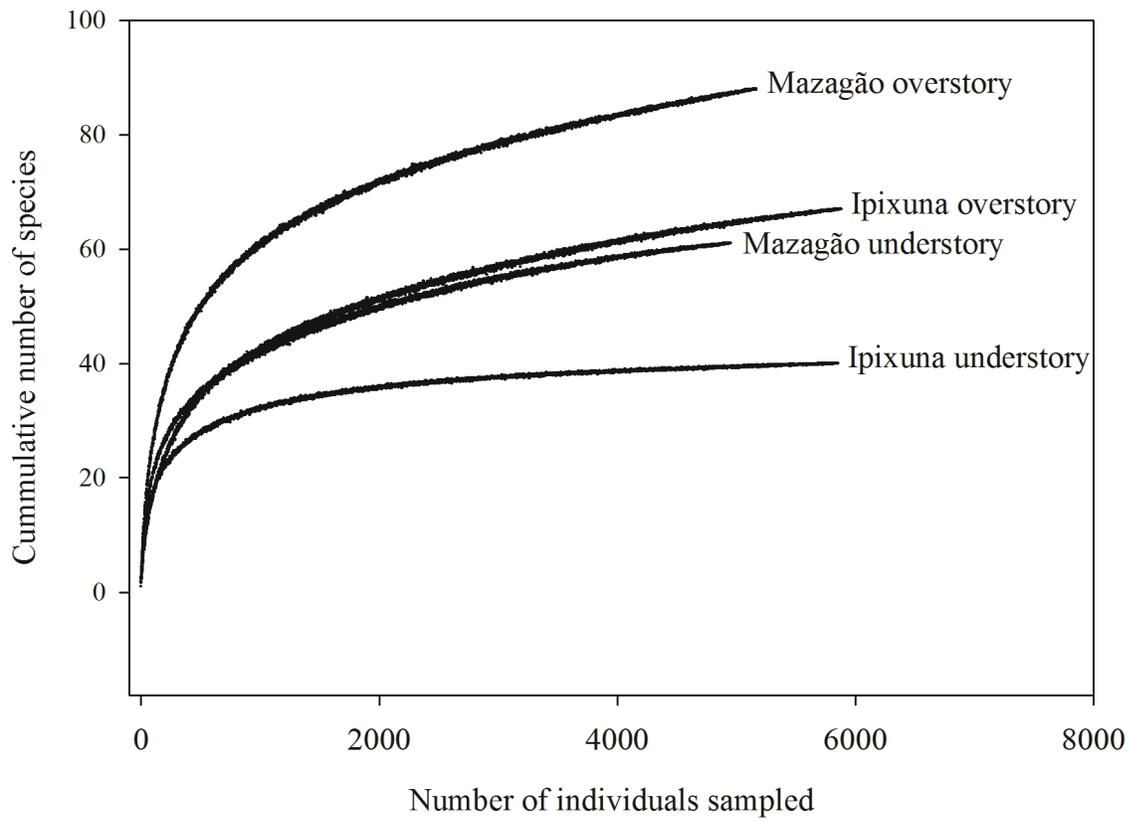


Figure 2-3. Species accumulation curves for overstory and understory of Mazagão and Ipixuna sites.

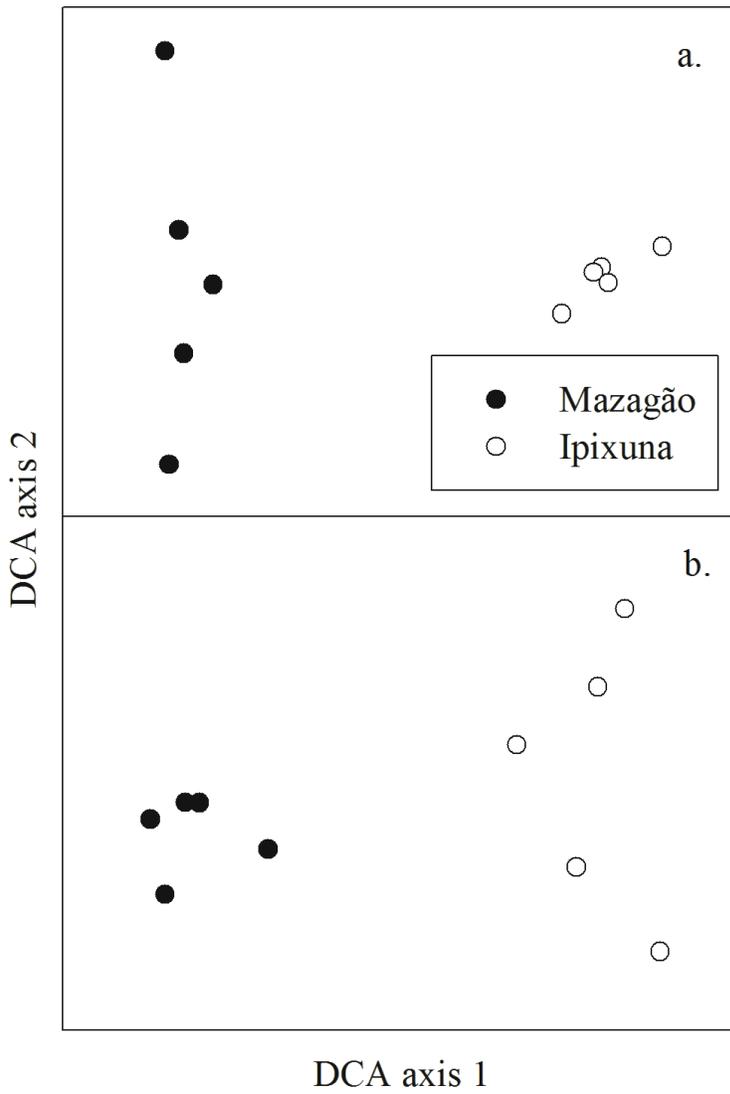


Figure 2-4. DCA results for (a) overstory and (b) understory for Mazagão and Ipixuna.

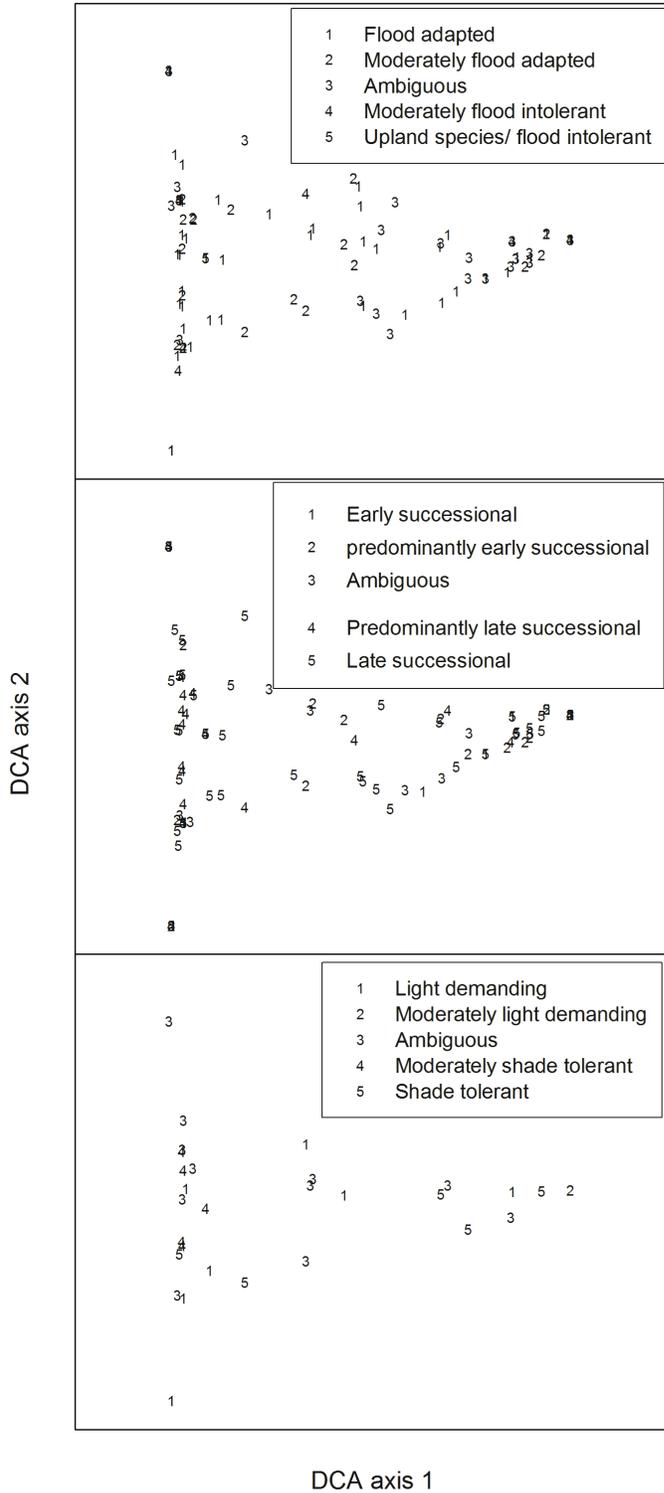


Figure 2-5. Individual species in DCA ordination space with respect to species characteristics.

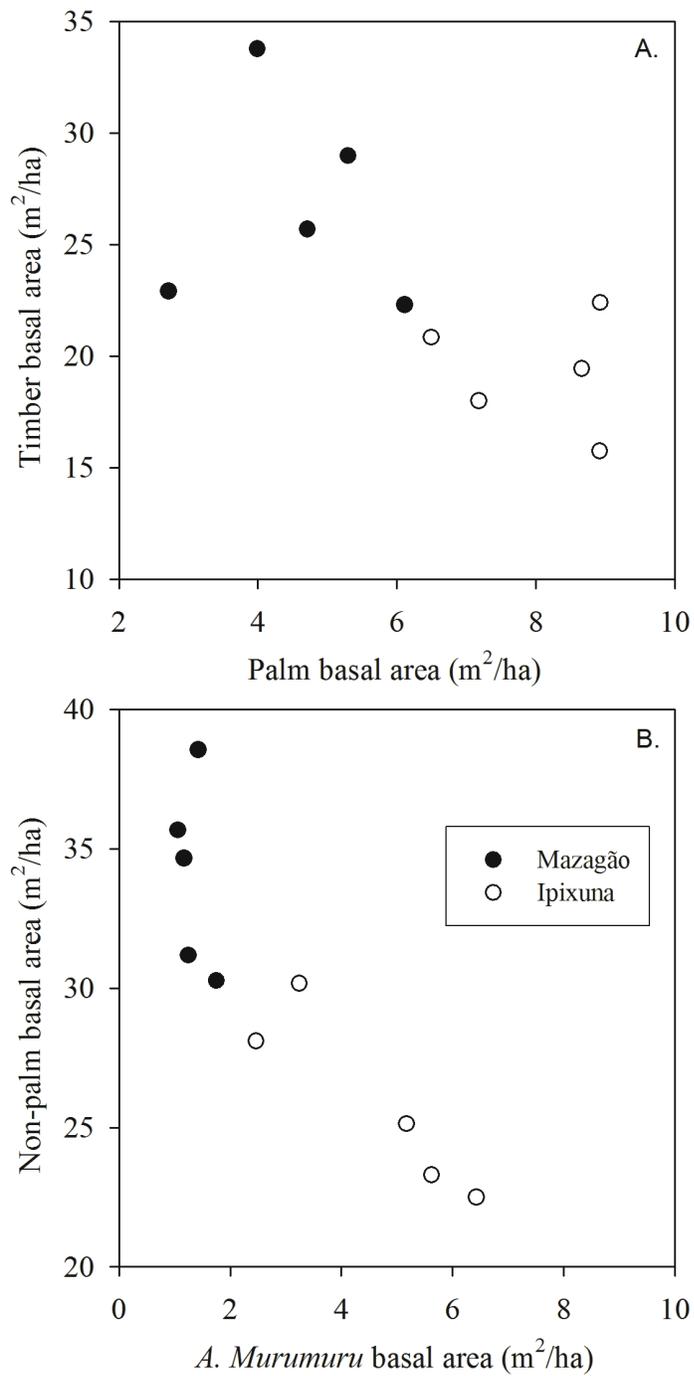


Figure 2-6. Negative relationships between palm and non-palm dominance Timber basal area versus palm basal area (A) and non-palm basal area versus *A. murumuru* basal area (B).

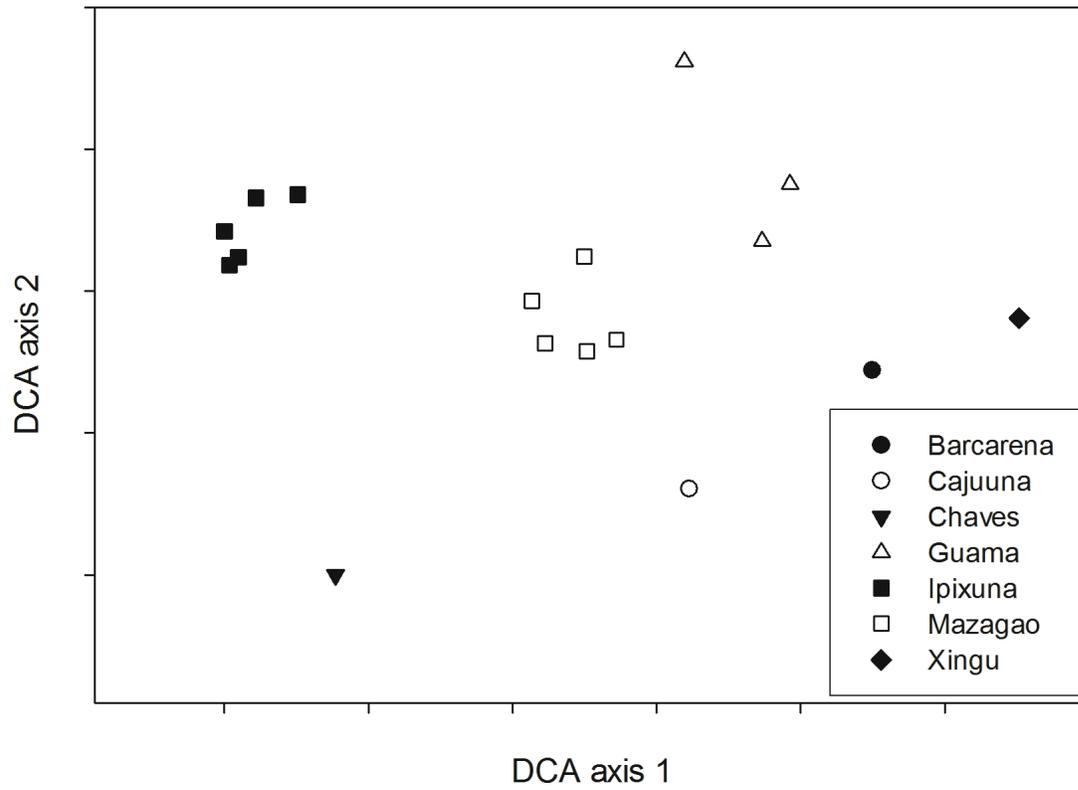


Figure 2-7. Two-dimensional DCA solution for overstory data (DBH > 10 cm) from Mazagão, Ipixuna and Guamá at the genus level.

CHAPTER 3
POPULATION DYNAMICS AND MANAGEMENT OF AMAZON TIDAL FLOODPLAIN
FORESTS: LINKS TO THE PAST, PRESENT AND FUTURE

Background

Amazon tidal floodplain forests were among the first of the Amazon's resources to be exploited by Europeans and represent the oldest commercial logging frontier in the Amazon (Barros and Uhl 1995, Raffles 1999). Floodplain forests that today are classified as intact often hide a past of crop and cattle production and a long history of timber extraction (Hiraoka 1992, Denevan 1996, Raffles 1999, Roosevelt 1999).

Until the mid 20th century, the post-contact use of tidal forests for timber was a highly selective high grading of a handful of species for log exports (Pinedo-Vasquez et al. 2001). Tidal forest timber harvests dramatically accelerated in pace and intensity from the 1950s until the 1990s when industrial-scale plywood and sawmills consumed a large volume of tidal forest wood from a wider selection of species (Palmer 1977, Browder 1989). As the supply of large high-valued trees decreased, these large estuarine mills were gradually replaced by small-scale family-run mills (Pinedo-Vasquez et al. 2001). Today, thousands of families supplying hundreds of small-scale timber operations rely on timber as a source of income (Lentini et al. 2005) by harvesting tidal floodplain manually, selectively, and without any fixed cutting cycle.

Past research on estuarine timber production has documented cases of apparently sustainable forest use in the region (Pinedo-Vasquez et al. 2001, Sears and Pinedo-Vasquez 2004), while others have argued that timber resources are being depleted (Macedo and Anderson 1993, Barros and Uhl 1995). Given the long history of timber use in the Amazon Estuary, such retrospective analyses are necessary to better understand the prospects of forest management. However, past assessments lack the

quantitative analyses of species ecology and demography that are crucial to accurately evaluate claims of ecological sustainability or resource depletion. Furthermore, the absence of tree demography data still prevents a shift towards ecologically-based management after 300 years of unregulated timber extraction.

In this study we evaluate whether the demography of tropical tidal floodplain trees is compatible with long-term sustainable timber management, while considering the importance of past timber use in defining future prospects of forest management. We characterize species-specific abundance, size-class frequency distributions, spatial distributions, and recruitment, growth and mortality rates of timber species in tidal floodplain forests. We also employ matrix population models to explore the underlying population dynamics of common tidal floodplain species and explore the link between species population ecology and past, present, and future timber use and availability. In our analysis, we combine conventional tree demography methods with data from interviews with tidal floodplain *Caboclos* regarding land use and ecological knowledge.

Methods

Study Region

We conducted our research in the 160 sq km Mazagão watershed at the western side of the Amazon estuary (Figure 3-1). The Mazagão watershed has a long history of timber use (Pinedo-Vasquez et al. 2001) with current small-scale timber extraction as part of diverse livelihood strategies that often include palm fruit and timber harvesting, fishing, and cropping. Mazagão is similar in composition and land use history to several adjacent watersheds, as confirmed by region wide inventories and surveys conducted in 2005 (Fortini et al. 2006). Mean annual temperature is 27°C and average daily temperature varies by less than 3°C from month to month. Mean annual precipitation is

2550 mm and occurs mostly in the wet season months of January-May. This part of the Amazon estuary is characterized by freshwater tidal fluctuations of 2-3 m. Tidal variations including daily, bimonthly, and equinoctial fluctuations, coupled with seasonal river level variations due to precipitation, result in a localized gradient of flooding that includes microsites flooded 1-2 times a year, 1-2 times a day, or even constantly during the wet season. These high frequency flooding regimes shape community composition of these forests (Cattanio et al. 2002) and produce a dynamic forest landscape from rapid erosion and deposition (Allison et al. 1995, Wittmann et al. 2004).

Species Selection

Based on interviews we conducted with owners of the estuary's family-run logging operations and smallholders in 2003-2005, we chose 9 timber species that account for the majority of commercial volume extracted in the region in the past and present or that are expected to have timber value in the future (Table 3-1). Initially *Olmedia calouneura*, *Symphonia globulifera*, *Cedrela odorata*, and *Aspidosperma desmanthum* were also selected but were later excluded due to small sample sizes. Based on five 1 ha plots, monitored species represent 47% of stand basal area and 53% of eudicot basal area.

Cedrela odorata, *Platymiscium filipes*, *Carapa guianensis*, *Olmedia calouneura* and *Virola surinamensis* were extracted during the industrial scale logging boom that lasted from the 1950s until the 1990s (Pinedo-Vasquez et al. 2001). Since then, *Callycophyllum spruceanum*, *Carapa guianensis*, *Virola surinamensis*, and *Licaria mahuba* have been the primary species utilized by smallholder sawmills spread across the region. *Mora paraensis*, a species with high-density wood, has only been harvested

at commercial levels by caboclo smallholders in the last decade because its heavy weight and low buoyancy make ground and water transport difficult.

Permanent Inventory Plots

Species demography in recently undisturbed forest was estimated from 3 large-scale 360 x 360 m plots (13 ha per plot, 39 ha total; unharvested plots *hereafter*) established and monitored yearly from 2005 to 2008. We measured all trees ≥ 5 cm DBH in the 200 x 200 m core area of each large plot. Only trees ≥ 30 cm DBH were measured in the remaining area. Additionally, we measured all trees ≥ 5 cm DBH in five 1ha plots first measured in 1997 yearly from 2004 to 2007. We selected areas indicated by land use history surveys to have had no or very low intensity extraction in the recent 20-year past. To be broadly representative of forest stands elsewhere in the Amazon Estuary, our intent was to find stands that have a continual history of timber use but would likely not be under recent post-harvest effects (Silva et al. 1995, Boot and Gullison 1995, Finegan and Camacho 1999).

For all measured trees in our unharvested plots, we collected information on individual and environmental factors that could influence demography including flooding and light regime (Table 3-2). Commercial height (i.e., height of crown base) was estimated using a vertical hypsometer and ocular estimates frequently calibrated by hypsometer measurements. Study trees were tagged at 1.2m height or 50 cm above buttressing and stem defects, with tag height recorded. Annual tree measurements consisted of 2 diameter measurements 10 cm above and below tag height. The short distance between measurements allowed for measurement comparisons that improved data quality. Tree diameter increments were calculated using the average increment at both measurement heights. Additionally, in 2007 a random subset (approx. 5%) of all

trees was re-measured to evaluate measurement error, yielding a typical 0-1 mm error estimate that rarely exceeded 2 mm. A 5 m collapsible aluminum ladder was used for all inventories due to the high level of buttressing of many floodplain trees (Parolin et al. 2004). All trees with buttresses or defects extending above 6 m had diameter at breast heights (1.3 m; *hereafter* DBH) estimated. These estimates were not used for diameter increment calculations, but were still useful for diameter distribution and survival estimates. Field tests showed diameter estimates consistently within 5 cm from actual size due to the field crew's familiarity with size estimation used for selling and purchasing of logs.

Evaluation of Timber Use History

We used a combination of unstructured interviews along with structured surveys with sorting and participatory mapping exercises to evaluate the extent and intensity of past timber extraction in the watershed, changes in forest structure and composition due to past extraction, and the extent of local knowledge of timber species ecology and distribution. Approximately 30 participants were included in these activities; all were long-time residents that witnessed and directly participated in the extraction of timber resources since the 1970s. We used a snowball sampling approach for participant selection.

Analysis

All inventory data were stored in a single database where for each tree, annual measurements were preceded by columns specifying measurement quality (Table 3-3). This eased processing large amounts of individual tree data for multiple analytical purposes. The developed matrix model creation and analysis routines can be applied to similar datasets and are available by the author upon request.

DBH estimates

One known consequence of flooding is the high incidence of buttress formation in trees (Parolin 2002). This results in the absence of direct DBH measurements for a large portion of trees > 30 cm DBH. To obtain a DBH estimate for these trees, we extrapolated taper derived from all pairs of diameter measurements spaced 20 cm apart. To avoid using individual taper which varied from tree to tree (and was sometimes negative for the measured stem section), we employed a multiple linear regression approach to model taper as a function of tree diameter and height of measurement by species. If the model used for a given species resulted in poor fit, we used generic coefficients from all species combined. We then used this species-specific taper model to extrapolate diameter from measured height to breast height (1.3 m). Although this method ignores the Neiloid shape of lower stem sections (Husch et al. 1972), extrapolated distances were mostly less than 1 m. After generating DBH estimates for all trees, we used field data to model species-specific commercial height/DBH relationship using a power function (Eq. 3-1).

$$Height = b \times DBH^a \quad 3-1$$

For species with height models not statistically significant, a general model using all species was used. Significant parameter estimates varied widely among species (mean $a = 0.43$ $sd = 0.31$; mean $b = 4.62$ $sd = 3.78$).

Estimating commercial volumes

Since there were no species-specific bole volume equations for all our study species, we used a commercial stem volume equation derived in Mazagão for *C. spruceanum* to all of our study species (Eq. 3-2; Applegate et al. 2000).

$$\text{Commercial stem volume} = (1.93 \times 10^{-4}) \times \text{DBH}^{1.665} \times \text{Height}^{0.973} \quad 3-2$$

Using a limited field-derived diameter and stem volume data set (Johnson 1999), we compared the quality of the fit of the above equation to other generic Amazon derived bole volume equations (Brazil 1978, Ribeiro 1996). Given the similarity in form of floodplain trees, the *C. spruceanum* equation showed a better fit to the field data.

Create population models for species or species groups

To evaluate the population dynamics of the study species, we created matrix models based on lower-level vital rates (Table 3-4, Morris and Doak, 2002). All species-specific matrix models are based on modeled tree growth, survival, and recruitment as described below. For maximum analytical flexibility, matrix model creation, subsequent projection, and analysis routines were all directly linked to the inventory database. This approach allowed for flexible outputs based on user-specified DBH size classes, plot selection, species subselection or grouping, and multi-year and cross-year averaging and bootstrapping.

Modeling diameter increments and matrix transition probabilities

To determine the probability of up growth from one size class to another we first used an increment estimator approach based on the mean increment distance from the upper size class boundary, appropriate for small sample sizes (Picard et al. 2007). However, to avoid over-parsing the data into a limited number of size classes, we opted for modeling diameter increment (ΔDBH) by tree size (DBH) using the Hossfeld IV equation (Eq. 3-3; Zeide 1993, Zuidema and Boot 2002). Comparison of same-sized matrices derived from parsing tree data into individual size classes and our equation-based methods yielded nearly identical results.

$$\Delta DBH = \frac{b \times c \times DBH^{(c-1)}}{[b + (DBH^c / a)]^2} \quad 3-3$$

Beside using increment data for matrix model parameterization, we used this information along with auxiliary data collected for each tree including estimated flooding regime, vine loading, and light regimes in ANOVA and standard least square models to evaluate the influence of these factors on diameter increments. Flooding effects were not evaluated for *C. spruceanum* and *P. filipes* because they had few trees in the highest elevation class. Statistical significance for all statistical tests was defined as $\alpha \leq 0.05$. Marginal significance was defined as $0.10 \geq \alpha > 0.05$.

Modeling survival

For all species and years of interest, we estimated size class survival using a binomial logistic fit (Morris and Doak 2002). The derived mortality function was then evaluated at the median observed size for each size class for each species/ year combination. If the logistical fit was not statistically significant, the matrix creation routine uses standard mortality rates calculated across all individuals (i.e., proportion of trees surviving a given time interval). In few species/ yr combinations, low mortality rates and numbers of trees resulted in the survival of all individuals. In these cases, the matrix creation routine used generic mortality rates curves derived from grouping all species combined for the same time period. Lastly, if calculating survival rates for intervals > 1 year, the matrix creation routine discounted rates to a yearly rate using a simple rate discounting function.

Estimating recruitment and modeling fertility

Fertility estimates were based on observed recruitment to the smallest size class (5 cm DBH) observed yearly at permanent inventory plots. Fertility was distributed

among size classes using Caswell's (2001) approach to calculate anonymous fertility rates. Fertility was distributed among size classes using the reproductive weight of individuals per size class, calculated as relative individual reproductive output times proportion of reproductive individuals in each size class.

While size is likely to influence reproductive output, the relationship is likely not linear due to increased respiratory costs with increasing tree sizes (Mencuccini et al. 2005). On the other hand, light limitation is likely an important threshold to reproductive output (Svenning 2002). To calculate relative reproductive output we devised a combined approach where the size class of full reproductive output is assumed to be reached when most trees reach full crown illumination (based on data gathered for each tree). Trees in this or larger size classes have full reproductive weight, whereas smaller size classes have relative reproductive output based on relative size compared to the size of the first full reproducing size class. To estimate the proportion of reproductive individuals for any given size class, in 2007 we employed a local expert to evaluate each tree growing in PIP plots for their potential to flower and reproduce. The matrix creation routine uses this data in a logistic regression by chosen species or species group to estimate the proportion of reproducing individuals by size class. Using this modified anonymous fertility approach, size at first reproduction was > 20 cm for all species.

To avoid problems associated with reducible matrices (Caswell 2001), in the instances we did not observe recruitment for a species group and time interval, we used recruitment values observed between 1997 and 2005 at the five 1 ha plots, discounted for annual mortality rates. Preliminary comparisons showed that while this ad-hoc

reducibility fixed avoided erroneous right and left eigenvectors, it had only minor effects on estimates of population vital rates and population growth.

Exploring tree population demography with matrix models

Resulting matrix models allowed me to calculate population growth rates (λ), sensitivities of λ to vital rate changes, and stable age distributions (Cochran and Ellner 1992, Caswell 2001) to explore the demography of tidal floodplain timber species. To avoid λ inflationary effects from small matrices (Zuidema et al. 2010), most size classes were set 2 cm wide. This value was the minimum width above maximum yearly individual growth during the entire study period. Model results were consolidated into larger size classes to aid in interpretation. We excluded temporal growth autocorrelation from our models after preliminary model projections deemed its effect negligible (Brienen et al. 2006). Additionally, we calculated population growth sensitivity to logged individuals and logged volume by expanding on the formula for sensitivity of population growth rates to size class survival rates (Eq. 3-4; Morris and Doak 2002).

$$\frac{\partial \lambda}{\partial s_j} = \sum_{i=1}^s \sum_{j=1}^s \frac{\partial \lambda}{\partial a_{ij}} \times \frac{\partial a_{ij}}{\partial s_j} \quad 3-4$$

Sensitivity of λ to volume of individuals harvested from a given size-class can be estimated based on a simple expansion using the chain rule (Eq. 3-5).

$$\frac{\partial \lambda}{\partial m_j^3} = \sum_{i=1}^s \sum_{j=1}^s \frac{\partial \lambda}{\partial a_{ij}} \times \frac{\partial a_{ij}}{\partial s_j} \times \frac{\partial s_j}{\partial I_j} \times \frac{\partial I_j}{\partial m_j^3} \quad 3-5$$

$\partial S/\partial I$ is the sensitivity of survival rate of a given size class to the additional death (e.g., harvest) of an individual of the same size class; and $\partial I/\partial m^3$ represents the ratio of individuals by cubic meter of wood. This approach can help find harvest strategies that

avoid depleting all large trees that typically have large contributions to population growth but relatively low densities (Zuidema and Zagt 2000).

Results

Diameter Distributions

Mora paraensis, *L. heteromorpha*, and *P. sagotiana* have clear inverse-J diameter distributions that seem to reflect their shade tolerant recruitment (Figure 3-2). These same species, along with *C. spruceanum*, were represented by larger trees than the remaining species. *Carapa guianensis*, *V. surinamensis*, *L. mahuba* had more even diameter distributions, while *P. filipes* and *C. spruceanum* had unimodal size distributions with peaks around 35-45 cm DBH. Only *M. paraensis* and *C. spruceanum* had densities of harvestable trees > 1 tree/ha (15.7 and 1.5, respectively). *Mora paraensis* was more than twice as abundant as all other harvestable study species combined. DBH-height model results yielded good results for most study species. Resulting fits were used to improve the estimates of tree commercial volume. Harvestable volume for all trees > 50 cm DBH was 64.6 m³ ha⁻¹, 51 m³ ha⁻¹ of which was from *M. paraensis*.

Because monitored trees were growing in relatively undisturbed stands, we examined diameter distributions for indications of shade tolerance species groups (Hartshorn 1980, Swaine and Whitmore 1988). While the resulting groups were compatible with traditional ecological data collected in assessment of local ecological data, we opted not to group species while modeling (Grubb 1977) because species in each shade tolerance group differed in size-class specific growth and survival patterns (Grubb 1977).

Diameter Increment Analysis

Mean and median diameter tree growth for all species across the 3 measurement intervals was 0.32 and 0.25 cm/yr, respectively. Except for 60-74.9 cm *V. surinamensis*, mean diameter tree growth for all species across all size classes was < 1 cm/yr (Figure 3-3). Tree growth was most often < 5 mm/yr for most trees measured. *Mora paraensis*, *C. spruceanum*, and *L. mahuba* had the lowest mean diameter tree growth. *Licania heteromorpha*, *C. guianensis*, *P. sagotiana*, *P. filipes* showed intermediate rates. *Virola surinamensis* had clearly higher tree growth rates than all other study species.

Mora paraensis, *L. mahuba*, and *C. spruceanum* showed slow tree growth throughout their size ranges, with small differences between maximum and mean tree growth. All other species showed varied tree growth patterns across size classes with maximum increment generally following the same patterns as mean tree growth, except when small sample sizes clearly influenced outcomes (typically the largest size classes with few measured trees). While some species exhibited decreasing diameter increments within the largest size classes, this did not lead to zero or negative tree growth for any of the species. In contrast, *C. guianensis* and *V. surinamensis* showed increasing mean tree growth as trees reached maximum sizes. These increment patterns could be indicative that larger observed trees of these species do not represent the largest possible sizes due perhaps to past episodes of selective logging.

Callycophyllum spruceanum shows near zero tree growth in the smallest size class, possibly reflecting its limited success under a shaded canopy. *Licania heteromorpha* and *P. sagotiana* showed optimum tree growth in middle size classes with growth rates declining in smaller and larger trees. Species of similar shade tolerance, as estimated

by size class frequency distributions, did not show similar diameter increment patterns across size classes.

Tree Growth Determinants

Neither forest type nor vine load affected growth rates, but trees suffering high vine-loads or growing in gaps were uncommon. Additionally, plot effects modeled as random effects were not significant for any species, meaning that tree growth variation observed is not explained by plot locations. In contrast, both flooding and light clearly affected tree growth rates. Tree growth for all species except *L. mahuba* increased at least marginally with increased crown exposure. Flooding had at least a marginally significant negative effect on growth for *M. paraensis*, *L. heretomorpha*, *C. guianensis* and *V. surinamensis*. Although these results show the link between tree growth, and crown exposure and flooding regimes, these two factors combined explained only about 7 to 25 % of the variation within study species.

Survival and Recruitment Estimates

Survival rates across all size classes varied little for all species during the study period, with values between 0.980 and 0.993, and 0.988 when averaged across all species and years. No clear pattern of decreasing mortality with size class was observed for any species or species groups. Given the small range of survival rates and the apparently stochastic nature of mortality events, small differences among species are not easily interpretable.

There was an average of 0.006 new recruits per standing tree per year into the 5 cm diameter size class during the study period. Recruitment per area of *M. paraensis* was higher than all other study species combined (Table 3-5). *Callycophyllum spruceanum*, *L. mahuba* and *P. sagotiana* showed average recruitment rates < 3 trees

per 100 ha per yr for the entire measurement period. Considering recruitment relative to population size, recruitment rates for most species was between 0.5-2% per year. Although *P. filipes* had a markedly higher recruitment rate (6%), this only translates to approximately 13 trees recruited per 100 ha per year given low population densities. We evaluated the sensitivity of matrix models to the choice of fertility calculation method (conventional fertility spread evenly across size classes and our modified method) and found nearly no differences in model behavior (<0.001 change in λ). Crown illumination proportion and reproductive proportion, both used to scale fertility across size classes and calculated from local ecological knowledge, showed clear size-related logistical patterns (Figures 3-4 and 3-5).

Evaluation of Timber Use History

Participatory mapping and surveys reveal that no forests in the Mazagão watershed (and surrounding regions) have escaped logging over the past 60 years. Similar to other Amazon tributaries, the mouth of the Mazagão river is where past landowners enforced trade of products from upstream residents for outside goods at the 'barracão' (trading post; Raffles 1997). Besides agricultural products, rubber and other non-timber forest products, high-value logs from species such as *C. odorata* always figured prominently in the list of local products traded. Starting approximately in the 1950-60s, the volume of timber extraction increased as large ships bound to farther markets docked at the Mazagão trade post and purchased large volumes of high-quality *P. filipes* and *C. guianensis* logs. Allegedly to make use of lower grade logs, a medium-sized band sawmill was installed at the location, all the while large rafts of *C. guianensis* and *V. surinamensis* logs floated downstream during high tides to the large but short-lived saw- and plywood mills near the state capital. After their demise due to lack of

continuous timber supply for their high capacity operations, along with the failure of the Mazagão sawmill, several small-scale circular sawmills began to dot the estuary (Barros and Uhl 1995, Pinedo-Vasquez et al. 2001) with the number of such mills in the Mazagão watershed varying from 6-12 at any given time since then.

Nearly all long-term residents were involved in this history of timber use and recall the extraction of markedly larger logs than maximum tree sizes observed today for several species with long timber use history including *C. guianensis*, *V. surinamensis*, *P. filipes*, and *C. odorata*. Independent ranking exercises show that these were the species with most volume harvested in the region excluding *C. odorata*, which was ranked as one of the least abundant species approximately 40 years ago. Contrary to results from long-used species, maximum recalled harvested log sizes for *M. paraensis* and *C. spruceanum* were within the range of maximum tree sizes observed within and outside inventory plots. As partial cross-validation of results, the ranks of study species by recruitment rates offered by local participants differed little from plot-derived results. Locals also correctly ranked *V. surinamensis* as the fastest growing species, but were not as accurate in ranking tree growth rates of most other species.

Matrix Model Outputs

Population growth estimates

Most species displayed bias-corrected bootstrapped λ confidence intervals including 1 across the 4-year measurement period, indicating the possibility that these populations are stable over longer time intervals (Figure 3-6). Only the fast-growing *V. surinamensis* and *C. guianensis* exhibited λ s clearly above 1 for 2007-2008, while both *L. mahuba* and *C. spruceanum* exhibited λ s significantly below 1, indicating potential

population declines. Population growth rates seemed to increase during the last measurement interval of 2007-2008.

Stable λ recruitment rate

Given the likely stochasticity of recruitment events and their potential consequences to modeled population dynamics, we devised a simulated-annealing-like search algorithm to find the recruitment rate at which populations are stable ($\lambda=1$). At each iteration, the search window narrowed until the calculated λ was within 10^{-9} of 1. Although the absolute differences between observed and stable population recruitment rates were small for all species, proportionally these differences were very large for *L. heteromorpha*, *C. spruceanum* and *L. mahuba* (Table 3-5). For these populations, poisson distribution tests show that measured recruitment rates are unlikely statistically different from stable population recruitment rates.

Stable stage distributions and reproductive values

The observed diameter distributions of species least harvested in the past follow those derived from stable stage distributions (*M. paraensis*, *L. heteromorpha*, and *P. sagotiana*; Figure 3-7). All other species show some form of deviation between actual and stable state distributions. For several species, this discrepancy was largest for the larger sized classes. *Callycophyllum spruceanum* and *L. mahuba* showed strongly bimodal stable stage distributions that were likely a result of nearly no observed recruitment and resulting population decline. *Platymiscium filipes* also showed large discrepancies between actual and stable state distributions, but it is not clear why. For all species but *M. paraensis* maximum reproductive values are reached between 34 and 44 cm DBH and remain high until maximum size. *Mora paraensis* shows the same

pattern of increasing reproductive value with size, but only reaching a peak at the 54 cm DBH.

Population growth sensitivities and elasticities

Computed elasticity matrices show the typical pattern of long-lived, slow-growing species where λ is most sensitive to stasis probabilities; However, most species show gradually decreasing sensitivity to stasis probabilities with increasing size, excluding the stasis probabilities of the largest size classes that were clearly important for most study species. The analysis of population growth sensitivity to underlying vital rates showed this pattern was due to the overwhelming influence of survival to population growth, which followed a similar size-related pattern as described above. Survival of large trees was a strong determinant of *C. spruceanum* and *L. mahuba* λ s. With nearly no recruitment, the rate of population decline of these two species is strongly dependent on the persistence of large trees. In contrast, *M. paraensis*, *L. heteromorpha* and *P. filipes* λ s were especially sensitive to survival of the smallest size classes. Tree diameter growth rates contributed little to λ s. For all species, most of the limited contribution of growth rates to λ s was derived from growth of the smallest size classes.

Population growth sensitivity to harvested individuals and volume

For all species except *P. filipes*, harvest of larger individuals is expected to cause the largest drop s in λ s. Depending on the underlying vital rates and observed diameter distributions, these adult harvest effects substantially influence λ s for *M. paraensis*, *C. spruceanum* and *C. guianensis*. For other species such as *L. heteromorpha*, *P. sagotiana*, and *P. filipes* the greatest potential harvest effects were observed for multiple large size classes. The sensitivity of population growth to volume extracted by size class yielded similar results; However, these results are only valid for the current diameter

distribution because they are based on the contribution of an individual relative to the total number of individuals in a given size class.

Stable age distributions

Stable age distributions, indicative of tree age at each size-class, varied widely among species (Figure 3-8). The median time needed to reach the stable age of individuals in the first harvestable class was as low as 67 years for *V. surinamensis* and as high as 190 years for *C. spruceanum*. *Mora paraensis* displayed a relatively high value of 143 years. Median stable age of largest observed size class varied even more, with low values for *V. surinamensis* (95 years) and *C. spruceanum* with median stable age at full maturity of 480 years. However, these values are not easily comparable because species had differing maximum sizes.

Discussion

Species Distribution and Densities

Although varzea low species richness has been commonly associated with more abundant species (Wittmann et al. 2006), most study species showed low abundances similar to those of upland species (Pitman et al. 2001, Schulze et al. 2008). Commercial timber volume of the forests inventoried was also similar to those from upland tropical forest sites (Verissimo et al. 1992, Barreto et al. 1998, Sist et al. 1998). Most of this volume was attributed to *M. paraensis* and does not account for stem defects and volume wasted in harvests (Valle et al. 2007).

The extent to which current population distributions have been influenced by prior harvests cannot be precisely determined, but timber use surveys and calculated stable stage distributions together yield some clues about the past. Comparisons between stable stage distributions and measured proportional size distributions suggest a larger

number of harvest-sized trees for most species would be present under unlogged conditions (e.g., *C. guianensis*, *V. surinamensis*). Moreover, local surveys indicate that the maximum log size of long-used timber species has decreased over the past decades. Because models cannot project or estimate the size distribution beyond currently observed sizes, stable stage distributions may underestimate actual shifts in population structure. Except for *M. paraensis*, the current size distributions and population densities of all Mazagão timber species constrain the management potential of small landholdings by limiting per-area yields and requiring time for recovery that often cannot be afforded by resident smallholders (Pinedo-Vasquez et al. 2001).

Tidal Floodplain Tree Demography and Dynamics

It has been hypothesized that high soil fertility of floodplains could lead to more rapid tree growth rates than in upland forests (Kvist and Nebel 2001). In fact, averaging across all species and sizes, the 0.32 cm/yr diameter tree growth is over 50% higher than the Amazon-wide upland forest conservative estimate of 0.14-0.2 cm/yr by Da Silva et al. (2002), but is still below rates from plantation and actively managed stands. In seasonal floodplain forests strong tree growth periodicity leads to cambium dormancy (Schongart et al. 2002) but, although we did not measure within-year tree growth variability, the lack of deciduousness and no clear growth ring formation on nearly all species suggest most tidal floodplain trees have year-round tree growth (Brienen and Zuidema 2005). Nonetheless, most study species presented a negative growth response to flooding, reinforcing the idea of tolerance, and not specialization strategies, to flooding stress (Kozlowski 1984).

While these results show flooding affects tree growth, these effects are hard to incorporate into management because they are not easily altered and are related to

topographical variation occurring at microsite scales due to the numerous and far reaching natural drainage canals present in tidal stands. However, light availability showed a stronger and more consistent effect across study species, and is a stand quality that can be managed through thinning (Guariguata 1999, Dolanc et al. 2003). Population λ s low sensitivity to tree growth and high sensitivity to survival indicate attempts to increase tree growth (such as improving light regime through thinning) may increase yield but will offer limited improvements to sustainability of harvesting regimes compared to management prescriptions that reduce mortality (e.g., reduction of residual stand damage). Nevertheless the fact that the importance of tree growth to population λ s was highly concentrated in the smaller size classes of all studied species make it a potential target for management intervention.

Given observed diameter tree growth, tree age estimates were within the published longevity estimates of 500 years or less with the light wood species *V. surinamensis* showing considerably shorter longevity (Worbes et al. 1992, Brienen and Zuidema 2006, Schongart 2008). Species variability in median age of the first harvestable size class (50 cm) illustrates the challenge to the current use of single harvest rotation cycles in tropical forest management (Schongart 2008). The unexpectedly high stable age estimate for *C. spruceanum* is likely due to its lowest diameter growth among all study species, a surprising result for a pioneer species that suggests its limited success in closed-canopy mature forests with small gaps.

While the measured tree survival rates are on the intermediate range of published values for other Amazonian sites (Lieberman and Lieberman 1987, Schulze and Zweede 2006, Sist and Ferreira 2007, Swaine et al. 1987), the high sensitivity of λ

to survival bodes well for tidal floodplain timber harvests given low residual stand damage (Chapter 4). Unfortunately, because the relative importance of adult and juvenile survival to λ varied widely among species, general management prescriptions are unlikely to be effective across species.

Recruitment rates into the 5 cm diameter class were also low for most species (Condit et al. 1999), and for all species combined represented only 74% of mortality over the 3-year study period. Similarly low recruitment rates from older (albeit smaller) plots and local survey ranking results indicate these low recruitment rates are not the result of temporal/stochastic variability. Relatively high *M. paraensis* recruitment is likely enhanced by the abundant advance regeneration stock commonly present in these forests (Rabelo 1999). The proportion of trees likely to reproduce by size class calculated from local ecological knowledge assessments showed a clear size-related logistical pattern that seems to reflect underlying reproductive biology (Read et al. 2006, Naito et al. 2008). These results suggest this approach may also be a good alternative to weighing anonymous tree fertility data when no other reproductive data are available.

Population Ecology and the Past and Future of Forest Use and Management

Considering the population ecology of the study species collected from inventory and survey assessments, the long legacy of forest use in the tidal floodplain forests clearly will influence the future of forest use and management.

According to social surveys, *M. paraensis* was already the dominant species prior to most timber extraction in the region. Due to difficulties in transport, *M. paraensis* was historically harvested at only low levels and for that has likely benefited from past timber use by reduced canopy competition and enhanced recruitment (Chapter 4). The fact that most other species showed markedly lower recruitment rates and that *M. paraensis*

commonly dominates the forest understory with advance regeneration also suggest that this species was better positioned to make use of small logging gaps (Rabelo 1999). Today, compared to most other timber species studied, the abundance and inverse-J population distribution of *M. paraensis* makes it a species potentially fit for management (Schulze et al. 2005, 2008). In comparison, for species like *C. spruceanum* and *C. odorata*, which inventory and survey data indicate as light-demanding pioneer species, recruitment limitation is likely not overcome by small selective harvest gaps (Chapter 4). The fact that *C. spruceanum* trees are usually found in size cohorts lend support to this idea and suggest that large disturbances such as changing geomorphology, large blowdowns, and ironically, ancient clearings and intense timber harvests likely contributed to the species success (Nelson et al. 1994, Condit et al. 1998). *Cedrela odorata* has the unfortunate distinction of having very low natural densities, resulting in a combination of traits that led to its near population collapse in the studied watershed.

Concluding remarks on Amazon Estuary tree population ecology

At first glance, tidal floodplain forests offer challenges to management similar to those seen in tropical forests elsewhere: low recruitment, tree growth rates that are often species-specific (Dauber et al. 2005, Rondon et al. 2009) combined with variable population densities and size distributions (Schongart 2008, Schulze et al. 2008). The long historical timber use in the region also limits the opportunity for small-scale management of several species that have declined in abundance over decades of low intensity extraction. Management of these and other species that occur at low densities will likely require efforts that cross smallholder property boundaries, a prospect that likely entails its own set of challenges different from large-scale timber management operations or management of community-owned land. Like most other tropical forest

studies evaluating the ecological constraints on forest use and management (Dauber et al. 2005, Gourlet-Fleury et al. 2005, Brienen and Zuidema 2007, Valle et al. 2007, Grogan et al. 2008, Rondon et al. 2009), these results are not encouraging. Nevertheless a full ecological evaluation should consider the potentially large effects of selective harvests and silvicultural treatments on tree growth and recruitment (Silva et al. 1995).

While timber extraction in the estuary has endured centuries, our results suggest long-term timber use does not necessarily entail sustainability. For the Mazagão watershed, its long timber use history has meant a gradual process of resource depletion due to the sequential removal of preferable timber species over time. If current practices are left unchanged, the prospects for long-term management are likely to decrease further as the densities of trees fall to levels that make management economically unattractive at smallholder scales. Fortunately, in Mazagão's case, the abundant presence of *Mora paraensis* may still provide the opportunity for sustained timber management while historically depleted species with strong recovery potential (e.g. *Virola surinamensis*, *Carapa guianensis*) can regenerate. These results contrast with recent optimistic studies that emphasize the potential of agroforestry projects in the estuary (Sears and Pinedo-Vasquez 2004, Sears et al. 2007b). While such agroforestry projects are not a substantial component of the current estuarine timber economy, they should be considered as potential ingredients for a strategy to ensure recovery of historically degraded forests.

Table 3-1. Study species.

Family	Species	Local name
Fabaceae	<i>Mora paraensis</i>	Pracuúba
Chrysobalanaceae	<i>Licania heteromorpha</i>	Macucu
Rubiaceae	<i>Callycophyllum spruceanum</i>	Pau mulato
Meliaceae	<i>Carapa guianensis Aubl.</i>	Andiroba
Myristicaceae	<i>Virola surinamensis</i>	Virola
Lauraceae	<i>Licaria mahuba</i>	Maúba
Sapotaceae	<i>Pouteria sagotiana</i>	Maçaranduba
Fabaceae	<i>Platymiscium filipes</i>	Macacaúba

Table 3-2. Auxiliary information collected for inventory trees.

Forest type	Tree stature
<ul style="list-style-type: none"> • Closed canopy forest • Forest gap • Gap edge • Stream/ river edge • Vine forest 	<ul style="list-style-type: none"> • Understory • Subcanopy • Canopy • Upper canopy • Emergent
Crown Damage	Stem damage
<ul style="list-style-type: none"> • None; only small branches broken • At least one main branch broken but canopy > 50% intact • ≥50 % of crown lost 	<ul style="list-style-type: none"> • None, low • Medium • High: visibly affecting individual performance
Crown illumination	Vineload
<ul style="list-style-type: none"> • None, low or medium lateral light • High lateral light but overhead shade; partial overhead light; partial overhead and substantial lateral light • Full overhead light; full overhead and lateral light 	<ul style="list-style-type: none"> • None; few vines present • Substantial vineload but canopy not significantly shaded by vine canopy • Canopy heavily infested by vines and substantial shading occurring
Flooding regime (using local typology)	
<ul style="list-style-type: none"> • Igapó (floods continuously in wet season) 	

Table 3-3. Categories of measurement quality for model data use
Tree data used for
diameter

Year tree status	diameter distribution and survival	diameter increment
Diameter measured	x	x
Diameter estimated	x	
Circumference at BH estimated	x	
Not 'born' yet		
Dead	x	
Error- discard high measurement	x	x

Table 3-4. Matrix projection model based on underlying vital rates.

	Szcl 1 _{t+1}	Szcl 2 _{t+1}	Szcl 3 _{t+1}	Szcl 4 _{t+1}	Szcl 5 _{t+1}	Szcl 6 _{t+1}	Szcl 7 _{t+1}
Szcl 1 _t	$s_1*(1-g_1) + F_1$	F_2	F_3	F_4	F_5	F_6	F_7
Szcl 2 _t	s_1*g_1	$s_2*(1-g_2)$					
Szcl 3 _t		s_2*g_2	$s_3*(1-g_3)$				
Szcl 4 _t			s_3*g_3	$s_4*(1-g_4)$			
Szcl 5 _t				s_4*g_4	$s_5*(1-g_5)$		
Szcl 6 _t					s_5*g_5	$s_6*(1-g_6)$	
Szcl 7 _t						s_6*g_6	s_7

s_1 survival for size class 1; g_1 transition probability of size class 1; F_1 fertility rate for size class 1

Table 3-5. Actual vs stable population yearly recruitment rates

	Actual recruitment ha ⁻¹	Stable population recruitment ha ⁻¹	Δ recruitment ha ⁻¹	Proportional Δ recruitment ha ⁻¹
<i>Mora paraensis</i>	1.280	1.443	0.163	0.13
<i>Licania heteromorpha</i>	0.180	0.605	0.425	2.36
<i>Callycophyllum spruceanum</i>	0.030	0.096	0.066	2.23
<i>Carapa guianensis</i>	0.178	0.131	-0.047	-0.26
<i>Virola surinamensis</i>	0.148	0.025	-0.123	-0.83
<i>Licaria mahuba</i>	0.010	0.252	0.242	24.18
<i>Pouteria sagotiana</i>	0.030	0.040	0.011	0.36
<i>Platymiscium filipes</i>	0.136	0.025	-0.111	-0.82

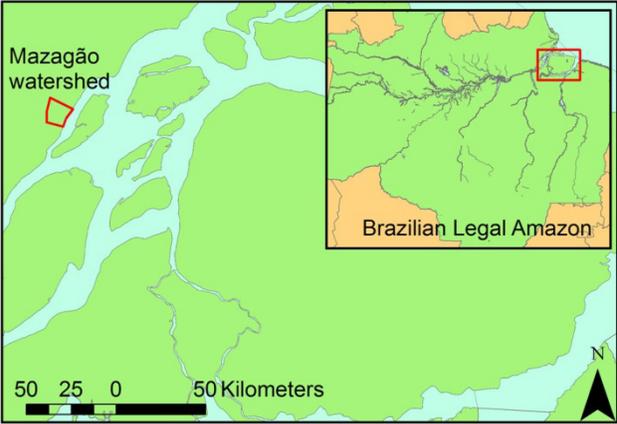


Figure 3-1. Study region

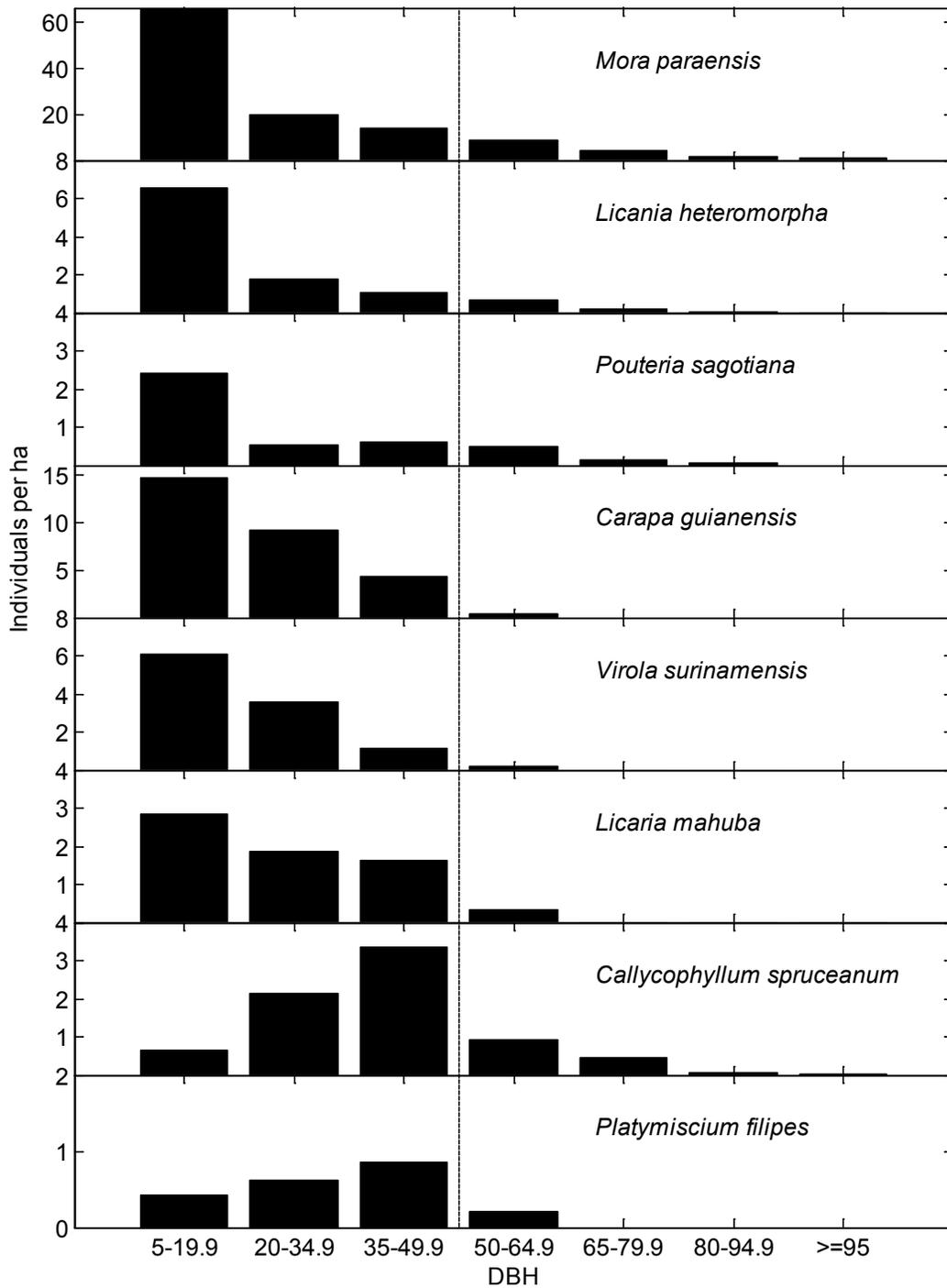


Figure 3-2. Diameter size distribution for studied species. Vertical dashed line indicates legal harvest size

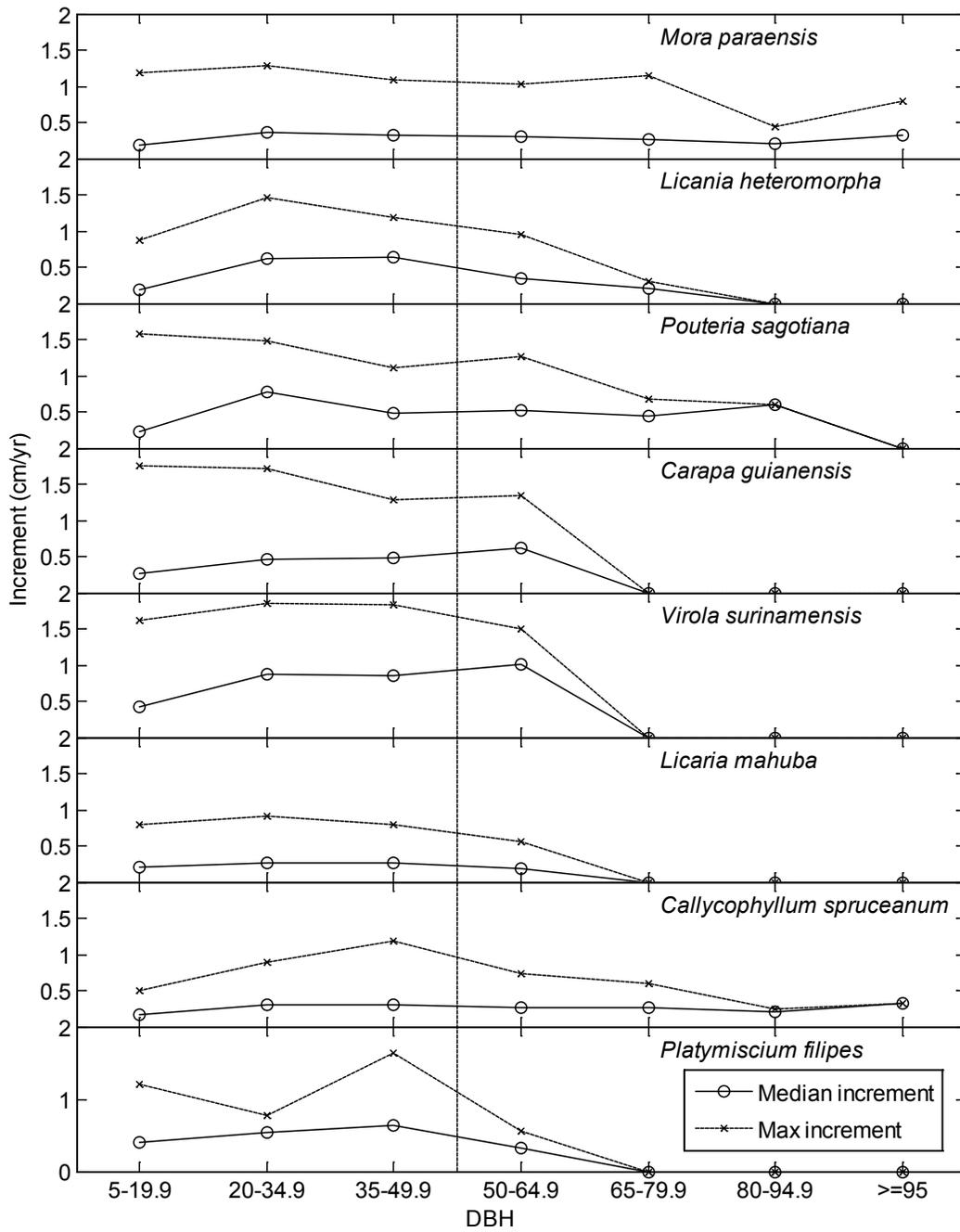


Figure 3-3. Median and maximum diameter increments. Vertical dashed line indicates legal harvest size

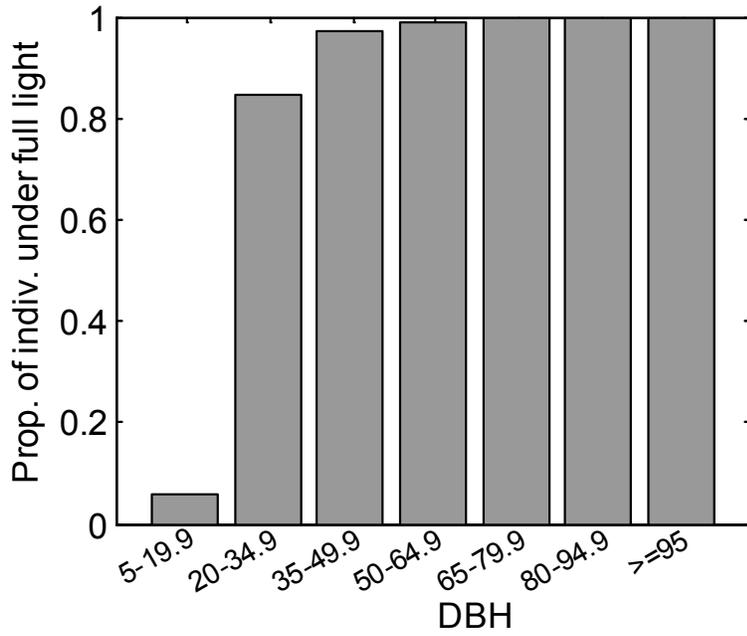


Figure 3-4. Proportion of trees under full light

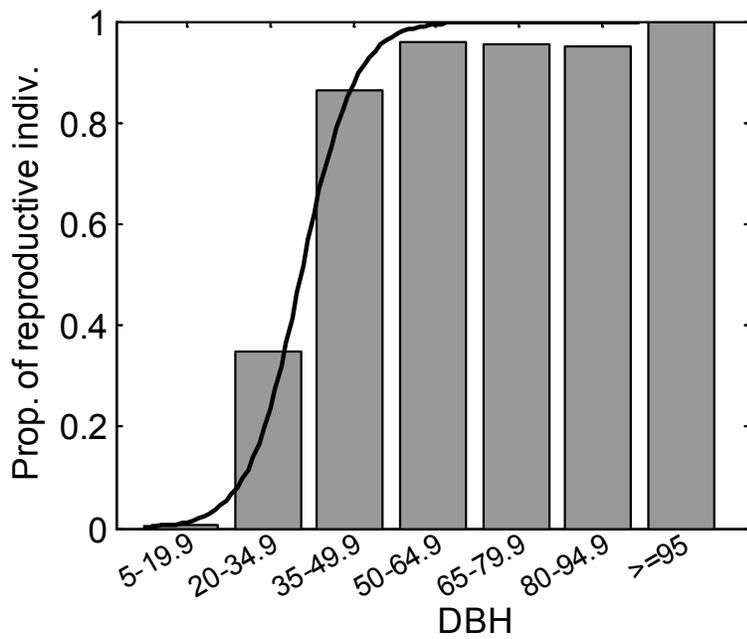


Figure 3-5. Proportion of reproductive trees

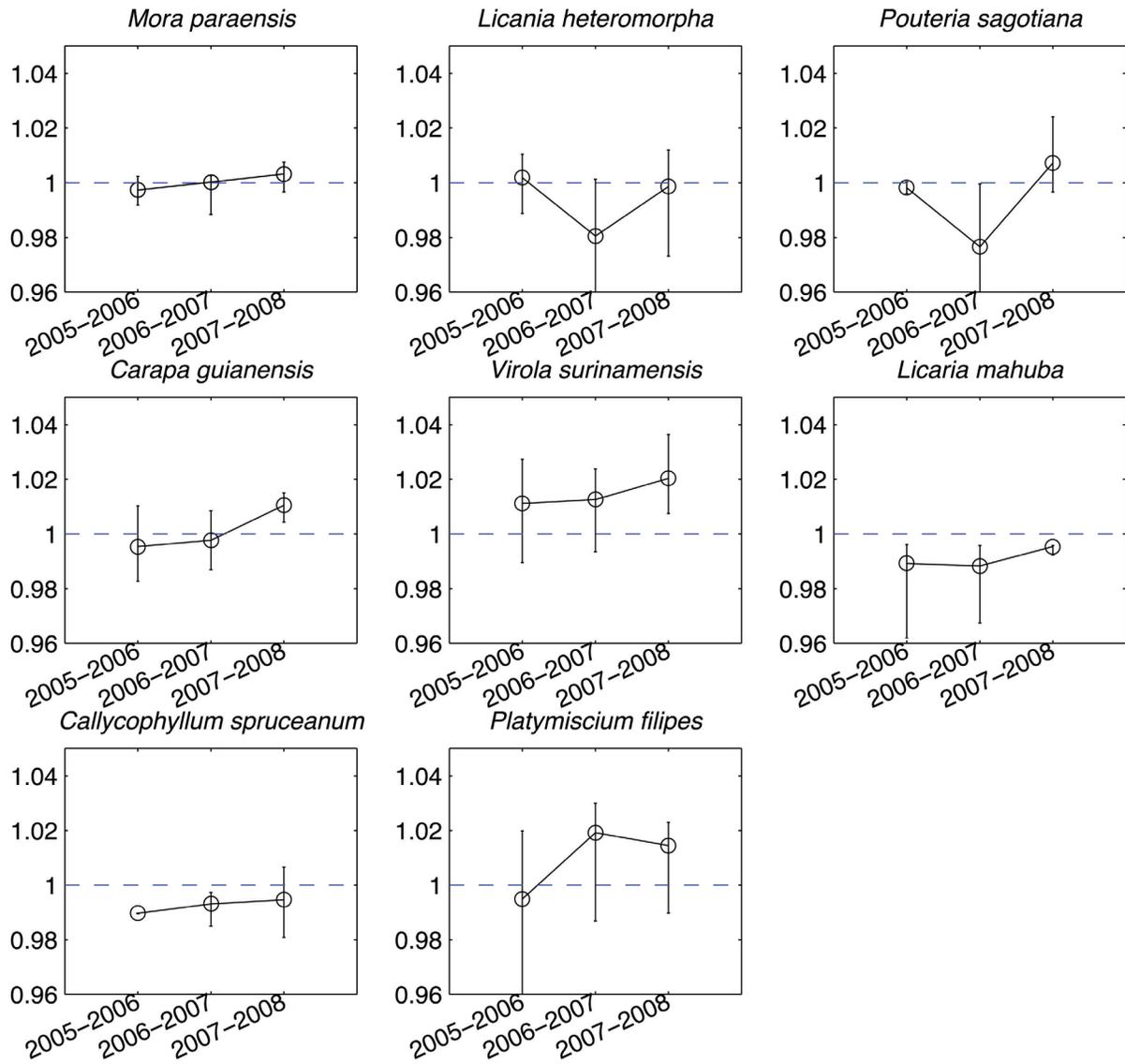


Figure 3-6. Population growth estimates for studied species by year with 95% confidence intervals

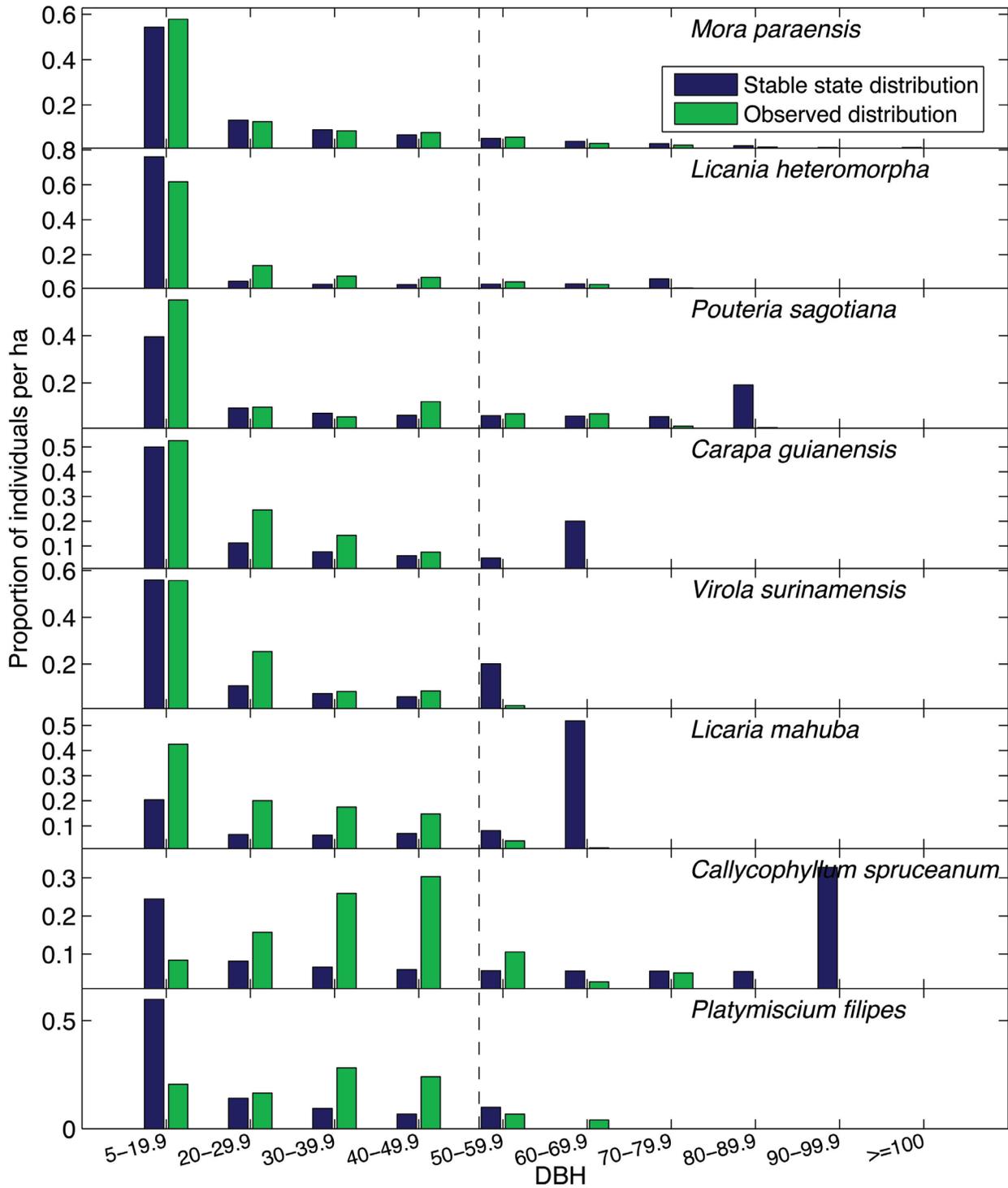


Figure 3-7. Observed vs stable state diameter distributions

Vertical dashed line indicates legal harvest size

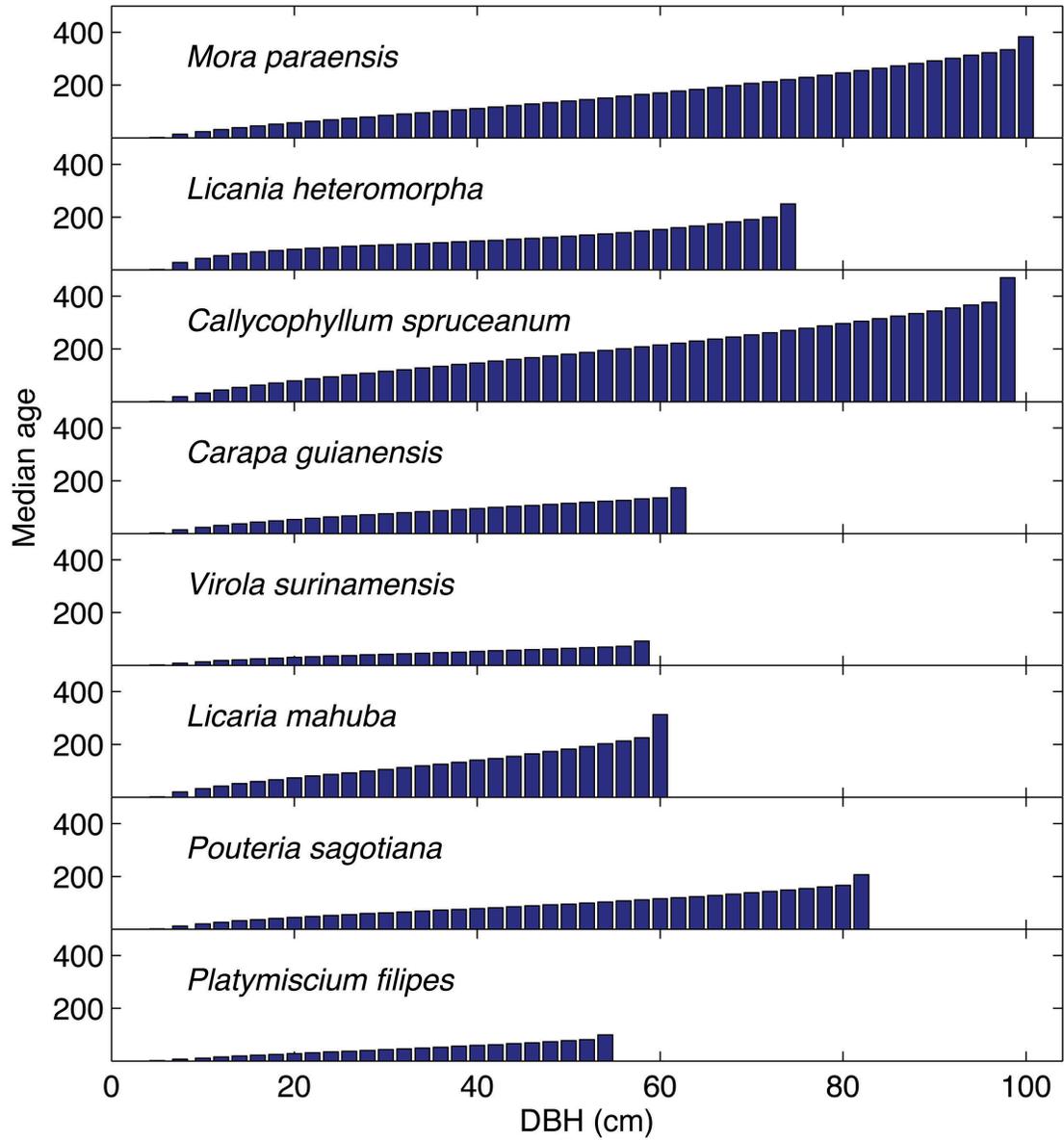


Figure 3-8. Median age at stable state distribution

CHAPTER 4 PROSPECTS FOR CONTINUED TIMBER PRODUCTION IN AMAZONIAN TIDAL FLOODPLAIN FORESTS

Background

One underlying tenet of sustainable forest management (SFM) is that forests should provide for multiple needs continuously into the future (Wang 2004). For remote tropical areas where timber is a main product, sustainable timber management (STM) to ensure yields harvest after harvest should be an important component of SFM, (Seydack 1995; Zarin et al. 2007 but see Luckert and Williamson 2005). While models have been applied to address the ecological viability of STM (Vanclay 1995, Kammesheidt et al. 2001, Glauner et al. 2003, Phillips et al. 2004, Gourlet-Fleury et al. 2005, Valle et al. 2007), most research has instead focused on the evaluation of harvest damage, post-harvest effects on growth, recruitment and mortality alone (Fredericksen and Mostacedo 2000, Chapman and Chapman 1997, Finegan and Camacho 1999). Without integrating these population demography responses, however, net population effects of timber harvest and management remain unclear. Exceptions aside (Huth et al. 2005), the majority of research to date has focused on evaluating current practices instead of searching for alternate sustainable management regimes relevant to concerned forest managers and legislators.

In the Amazon, the majority of STM-related research has resulted in discouraging prospects for both SFM and STM (Dauber et al. 2005, Gourlet-Fleury et al. 2005, Brienen and Zuidema 2007, Valle et al. 2007, Grogan et al. 2008, Rondon et al. 2009), but see Phillips et al. 2004). Nevertheless for the Amazon estuary, the oldest logging frontier in the region, lack of relevant research is a glaring omission. After centuries of disturbance, the resilience of these forests indicates a potential for sustainable timber

management (Raffles 1997). This potential is further bolstered by an abundance of timber, relatively fertile soil, and low cost/ low damage of water transport (Anderson 1990, Zarin et al. 1998). In relation to timber harvesting in Amazonian upland forests, floodplain forest operations have lower negative environmental impacts due to the lack of heavy machinery use and the reliance on river transportation that precludes road construction and eventual deforestation (Kaimowitz and Angelsen 1998, Laurance 2001). Additionally, the dangerous fire-feedback mechanisms that often hinder the recovery of upland forests following logging are absent in floodplain forests (Nepstad et al. 1999, Gerwing 2002, Cochrane and Laurance 2002).

While recent research explores the population ecology of timber species and its link to centuries of selective logging in the region (Chapter 2 and 3), it remains unclear to what extent tidal floodplain tree demography and past use affect the prospects for STM and SFM. At present, timber remains an important resource for subsistence and income generation for many smallholder *caboclo* families in the Amazon Estuary. Hundreds of small-scale, family-run timber operations supplied by many more thousands of smallholders rely on timber as a source of income (Barros and Uhl 1995, Lentini et al. 2005). While past research into these small-scale production systems have characterized *caboclo* timber use as either sustainable (Pinedo-Vasquez et al. 2001, Sears and Pinedo-Vasquez 2004) or unsustainable (Macedo and Anderson 1993). Attention to how demography dictates the possibility of recovery after management interventions has been lacking.

In this study we used a matrix-based harvest simulation model to evaluate the prospects for sustained timber production in the Amazon estuary. Based on a wide

range of values for management criteria (e.g., length of cutting cycle, minimum cut diameter), we projected the population and yield outcomes of hundreds of long-term timber management scenarios. These results were then compared using simple quantitative indicators of STM to find optimal stand-level and species-specific sustainable timber management regimes relevant to similar forests in the Amazon Estuary.

Methods

Study Region

We conducted our research in the 160 km² Mazagão watershed at the Western side of the Amazon estuary (Figure 4-1). This area has a long history of timber use (Pinedo-Vasquez et al. 2001) with current small-scale timber extraction as part of diverse livelihood strategies that often also include palm fruit harvesting, fishing, and small-scale agriculture. Despite great compositional and structural variability across Amazon tidal floodplain forests, Mazagão is similar in composition and land use history to several adjacent watersheds, as confirmed by region-wide inventories and surveys conducted in 2005 (chapter 3, Fortini et al. 2006). Mean annual temperature is 27°C and average daily temperature varies monthly by < 3°C. Mean annual precipitation is 2550 mm and falls mostly in January-May. This part of the Amazon estuary is characterized by freshwater tidal fluctuations of 2-3 m. Field data were collected primarily from recently undisturbed forests (forests with little or no sign of recent timber harvests but with a long history of multiple use including harvests) and recently logged forests.

Species Selection

We chose 9 timber species that account for the majority of commercial volume extracted in the region over the past century as well as species that are likely to have timber value in the future (Table 4-1). *Olmedia calouneura*, *Symphonia globulifera*, *Cedrela odorata*, and *Aspidosperma desmanthum* were initially selected but were later excluded because there were < 50 trees inventoried per species.

Cedrela odorata, *Platymiscium filipes*, *Carapa guianensis*, *Olmedia calouneura* and *Virola surinamensis* were extracted during the industrial scale logging boom that lasted from the 1950s until the 1990s (Pinedo-Vasquez et al. 2001). Since then, *Callycophyllum spruceanum*, *C. guianensis*, *V. surinamensis*, and *Licaria mahuba* have been the primary species utilized by small-scale sawmills in the region. *Mora paraensis*, a species with high density wood, has only been harvested at commercial levels by caboclo smallholders over the last decade because its weight and low buoyancy make ground and water transport difficult.

Permanent Inventory Plots

Species demography in recently undisturbed forest was estimated from 3 large-scale 360 x 360 m plots (13 ha per plot, 39 ha total; unharvested plots *hereafter*) established and monitored yearly from 2005 to 2008 and five 1ha plots first measured in 1997 and then yearly from 2004 to 2007. To evaluate demographic responses to logging, in 2007 we established 14 small permanent inventory plots (totaling 6.2 ha; harvested plots *hereafter*) in areas logged in 2001-2006. Due to the small scale of logging operations, harvested plots were either 60 x 60 m or 80 x 80 m to avoid surrounding unharvested areas. Using the same methodology for recently undisturbed

forest plots (Chapter 3), we measured all H plot trees in 2007 and 2008. A total of 5800 trees were monitored for this study.

In addition to the collection of demographic data from permanent inventory plots, for all measured trees, we collected information on individual/ environmental factors that could influence demography including flooding and light regimes (Chapter 3).

Commercial height (i.e., height of crown base) was also estimated for all trees from our unharvested plots using vertical hypsometers to improve tree volume estimates.

Estimating Population Commercial Proportion

To avoid the unrealistic yield-inflating assumption that all harvest-sized trees are commercial (Valle et al. 2006), all inventory trees were classified into commercial/ defective grade by a local field crew with expertise in tidal floodplain logging practices. We defined defective trees as those with defects severe enough that they would not be felled. Since past harvests altered the proportion of commercial trees in surviving populations, only data from seldom-harvested *M. paraensis* were used to calculate commercial proportion for all species. While species characteristics (e.g., physiognomy, wood density) likely influence the merchantable proportion of trees in undisturbed populations, the use of *M. paraensis* alone was preferred to avoid likely larger biases resulting from the inclusion of populations that have had their commercial proportion altered by past harvests. While this approach may lead to optimistic first harvests for species more heavily used in the past, it yields a clearer pattern of increased incidence of defects with size which is more important to projected long-term population dynamics. Furthermore, preliminary model results showed that species with long use histories contribute substantially less to first harvest yields than *M. paraensis*, making the size of the inherent bias in our approach likely small.

Monitoring of Harvesting Activities

In July – August 2008 we monitored the harvesting activities of 3 local logging crews to evaluate residual stand damage and related logging practices. A total of 413 man-hours were monitored in which the extraction of 40 standing trees was closely followed from forest to sawmill. We identified and measured the DBH of all trees harvested, damaged or killed during logging. By monitoring harvest effects during logging it was easier to detect damage and determine if it was caused by operations. Because damaged trees may not die immediately following harvests, trees with extreme harvest damage were assumed dead.

Harvest Model

To simulate the impact of differing harvest regimes on timber tree populations, we devised a matrix harvest model using methods described in Fortini (Chapter 3). The harvest model is an area-based, non-spatial model that allows for the specification of management criteria including harvest rotation length, multiple target species, minimum cut diameter (MCD), harvest intensity (1 - seed-tree retention proportion of trees with $DBH \geq MCD$), minimum commercial density (MD) and limits of harvest volume per ha. Model mechanics are described in detail below.

Modeling defective stems. We used a multi-site approach to model size-dependent shifts in commercial/ defective population ratios (Morris and Doak 2002). Commercial and defective portions of each species population were modeled separately (Table 4-2). Defective trees are still projected each year, may contribute to new regeneration, and may also suffer residual stand damage (*see below*). Based on the field data, the model assumes that for large-sized trees, a defective status is irreversible and that with increasing size, more commercial trees develop defects and

become unmerchantable. Harvests select only commercial trees (but also result in indiscriminant residual stand mortality) and thus result in a shift in the commercial/defective population ratios. Because the role of population genetics to the development of stem defects is unclear, projected shifts in the contribution of defective adult trees to population recruitment do not influence the probability of defects occurring in juvenile classes. Seed-trees selection can include the preferential selection of defective trees (low seed-tree quality, assuming short-term profit maximization behavior by harvester) or may include only the commercial proportion of the population (high seed-tree quality).

Modeling density dependence. We included density dependent recruitment regulation (Condit et al. 1994, Blundell and Peart 2004) using a complemented Weibull function (Haefner 1996) to avoid unrealistic model outcomes due to the intrinsic exponential nature of matrix projections (Bierzychudek 1999).

$$Recruitment_{Na} = a \times \exp\left(\frac{-Na}{b}\right)^6 \quad 4-1$$

Where a and b are parameters related to recruitment rates at average plot density (N_p) and at carrying capacity (K) and N_a is current stand density per ha. The complemented Weibull function was chosen because it regulates recruitment rates only when densities approach K , as opposed to other commonly used models (Cropper and Loudermilk 2006). This density-dependent approach avoids overestimates of population growth by not increasing recruitment rates when N_a falls below N_p . This was done by subtracting N_p from K during parameter estimation and calculating N_a by subtracting N_p from yearly population density estimates (but not allowing negative values). This normalization was used because all underlying demography (i.e., recruitment, growth,

and mortality) was by definition evaluated at N_p , with no auxiliary information to determine demography at lower population densities.

K was calculated as the maximum subplot population density observed in all data from our unharvested plot data using all tree sizes. This calculation was done through an embedded moving window algorithm with a 100 x 100 m window size. The ha-sized window not only was compatible with the model's per ha outputs but also was the smallest window size that avoided effects from small-scale variability in the distribution of trees. K and N_a in the model can be specified for individual species, user-defined density dependence groups, or all species combined. Since it is highly unlikely that the observed highest densities for all species would be observed in a same subplot due to limits on overall stand density, species-specific K s were scaled proportionally to the maximum density observed for all species combined. Species-grouped K and N_a model parameterizations allowed for the competitive replacement of species being harvested or dying naturally by individual of species with ample recruitment, allowing for a simple mechanism for species composition shift after logging.

Harvest damage modeling. Based on harvest monitoring field data, we calculated a residual stand mortality ratio. After each simulated harvest, we applied the ratio to all populations based on the number of trees harvested. Because there were no clear size-related patterns in residual stand mortality, the residual stand mortality ratio used in the model is indiscriminate of commercial and defective trees and size class.

Post-harvest demographic effects assessment and modeling. All trees surviving a harvest are modeled using a post-harvest matrix that includes post-harvest demographic effects. First, we attempted to define the length of post-harvest effects on

growth and mortality using standard least square regression of post-harvest diameter increments against years since logging. Unharvested plots were grouped for subsequent analyses since no effect of time since last harvest was found in the data. Post-harvest demographic effects were determined by the comparison of diameter growth, recruitment, and mortality rates between unharvested and harvested plots for the 2007-2008 measurement interval. These effects were included in projection matrices by considering the change in vital rates between the two treatments. To explain treatment differences in growth, we calculated the proportion of trees per species and size class under varying growth conditions (e.g., light, and stem and crown damage differences).

Management simulation model outputs. Every management simulation included a projection period of at least 120 years. Population growth rates (λ) under simulated harvest regimes were calculated empirically from model projections (Eq. 4-2).

$$\lambda_H = \left(\frac{N_{yr120}}{N_{yr0}} \right)^{(1/120)} \quad 4-2$$

Based on λ_H , for each harvest regime simulation, the model also calculates the necessary compensatory recruitment needed to stabilize population size at pre-harvest levels.

Simulation of multiple management regimes. We applied the management simulation model over a range of possible management criteria: cutting cycles of 10-40 years; MCDs of 30-70 cm DBH; harvest intensity of 0.5-0.9 of stand volume; minimum species density (MD) of 0.03 and 1 trees/ha per harvest species; high and low seed-tree quality (i.e., exclusion or inclusion of defective trees in the calculation of harvest

intensity); and simulations with or without a harvest volume limit of a $1 \text{ m}^3 \text{ ha}^{-1} \text{ yr}^{-1}$ (Table 4-3). A total of 1440 management regimes resulted from all possible management criteria combinations. The harvest model applied each management regime to all study species combined until the end of the 120 yr projection period. Model outcomes were also analyzed separately for each species to explore species-specific outcomes of common management criteria. The range of management regimes simulated also included the federal management guidelines as required by Brazilian law (i.e., 10 or 30 yr cutting cycle, maximum 90% harvest intensity, 50 cm MCD, MD of 0.03 trees/ha and a $1 \text{ m}^3 \text{ ha}^{-1} \text{ yr}^{-1}$ harvest volume limit; Table 4-3; <http://ibama2.ibama.gov.br>).

To evaluate which management regimes yielded the best long-term yield and population stability, all management regime simulation outcomes were rated according to 4 different STM indicators: annualized yield of fifth harvest ($H_5 \text{ AY}$), annualized yield of third harvest ($H_3 \text{ AY}$), mean annualized yield (mean AY) during the 120 years excluding the first harvest, and the combined rank sum of mean AY and λ_H (mean AY \cap λ_H). The $H_5 \text{ AY}$ indicator was considered indicative of STM since management regimes were applied by the model consistently across each harvest. This temporal consistency kept the model from 'cheating' by selecting low harvest intensities in initial harvests and higher intensities in later harvests. In contrast, a management regime chosen to improve future stocks that is excessively conservative will also restrict the volume of future harvests. The $H_3 \text{ AY}$ indicator was included to determine whether projection length influences analysis results. The mean AY \cap λ_H indicator was calculated simply as

the summation of the ranked mean annualized yield and the ranked λ_H . First harvests were excluded from mean AY because annualized yields are not computable.

To evaluate inter-specific differences in optimum management regimes, for each of the 4 indicators used, all optimal species-specific management regimes were compared. This comparison allowed the evaluation of how optimum management regimes for any given indicator varied among species. To determine whether differences in species-specific optimum management regimes led to suboptimal management at the stand level, we compared group and species-specific optimum management regime outputs. Lastly, we used correlation analyses to evaluate the influence of management criteria on simulation outcome indicators by harvest species combined and individually.

Results

Post-Harvest Demographic Effects

Based on comparisons between recently logged and unlogged plots, nearly all species showed a positive growth response after logging. Surprisingly, *C. spruceanum*, the most shade intolerant species in the study, was the only one that showed a small growth reduction in response to logging. Most species showed a proportional increase in growth rates of trees in the juvenile size class (5-20 DBH) and no consistent treatment response for the adult size classes (>20 DBH). For the purposes of including post-harvest diameter growth effects in management simulations, we maintained species response differences while summarizing post-harvest growth effects as juvenile and adult effects. Additionally, post-harvest diameter growth was not clearly related to maximum unlogged forest growth, precluding the use of maximum growth as post-harvest growth (Rondon et al. 2009).

While there were no large recruitment boosts into the 5 cm DBH class after logging for any focal species other than *Mora paraensis*, low recruitment and population densities precluded the possibility of observing small recruitment effects for the remaining species (i.e., demographic stochasticity is larger than expected range of recruitment increases). Given these results, we chose to model post-harvest recruitment increases only for *M. paraensis*. Post-harvest increases in mortality were not apparent for any focal species and were hence not included in the simulation model.

Overall, the comparison of population projection matrices using only 2007-2008 data for either unlogged or recently logged plots for *M. paraensis* and *C. spruceanum* (species with data sufficient to independently parameterize matrices for both treatments) showed that post-harvest λ increases are large for *M. paraensis* (1.0272 vs 1.0032; logged vs unlogged, respectively) but not for *C. spruceanum* (0.9944 vs 0.9946; logged vs unlogged, respectively). A life table response experiment decomposition of post-harvest effects by underlying vital rates (Caswell 2001, Morris and Doak 2002) shows that the increases in *M. paraensis* λ is mostly due to increases in growth of the smaller size classes and increases in fertility by the middle size classes, despite larger responses observed in the larger size classes (Figure 4-2).

There were no differences between unharvested and harvested plots in the proportion of trees suffering from stem and crown damage, or for vine loading, likely a consequence of generally low residual harvest damage. A larger proportion of 5-20 cm DBH trees were observed under high light conditions in harvested plots compared to unharvested plots (25% vs 7%, respectively). Using only data from harvested plots, an analysis of growth increment by time since logging revealed no clear temporal trend.

Based on these results and limited literature indicating that post-harvest effects on growth generally do not last longer than 10 years (Silva et al. 1995, Asner et al. 2004), we chose to use a post-harvest effect duration of 10 years. The difference in yield between using a 5 or 10 yr post-harvest effect scenarios was approximately $2 \text{ m}^3 \text{ ha}^{-1} \text{ harvest}^{-1}$, which does not drastically alter model projections and related conclusions.

Harvest Damage and Mortality

Observed residual stand damage was small and mostly affected palms and non-timber species (Table 4-4). Observed harvest-induced damage and mortality was minimal for trees $>35 \text{ cm DBH}$. For each m^2 of basal area harvested, an additional 0.11 m^2 of basal area from timber trees $>35 \text{ cm DBH}$ was removed due to damage and mortality from felling. Most notably, there was no observed damage or mortality of trees $\text{DBH} >35 \text{ cm}$ due to transport related activities.

Management Simulation Outputs

The model based on grouped-K density-dependence yielded simulation outputs in which the speed of shifts in species relative abundance seemed related to species λ s and intensity of management regimes (Figure 4-3). While using a group K allowed for the shift in species composition that is well documented in logging operations elsewhere (Dickinson et al. 2000, Fredericksen and Mostacedo 2000, Dekker and de Graaf 2003), this approach essentially yields a non-validated competition model. While such a minimalist model for stand dynamics merits future research, results presented herein utilize species-specific K s.

Despite the long history of timber use in the region, model outputs show that under the management requirements of Brazilian law, on average 50.9 m^3 of commercially harvestable timber is available per ha in the recently undisturbed forests

we sampled. Of this volume, 30 m³ would be available for harvest under a 30 yr cutting cycle and 10 m³ under a 10 yr cycle according to Brazilian law. Model projections indicate fast stock recovery allows continued extraction of maximum allowed volume in subsequent harvests. However, the harvest yields per species change considerably across harvests. Using the 30 year Brazilian legal scenario, *Mora paraensis*, which accounts for 87% of the first harvest, yields approximately 50% of the volume (15 m³) for subsequent harvests. Similarly, *Callycophyllum spruceanum* yields 9% of the first harvest volume but by the fifth harvest yields 4% of total harvest volume. On the other hand, yields of historically harvested *C. guianensis* and *V. surinamensis* increase from 2% (0.5 m³) during the first harvest to 23% (7 m³) during the fifth harvest for the two species combined.

Commercial proportion decreased for all harvested species due to harvesting in Brazilian 30-yr cutting cycle legal management regime without volume-based harvest limits (Figure 4-4). Nevertheless the rapid population turnover of species with high λ s results in the stabilization of commercial proportion at high levels before each simulated harvest (e.g., *V. surinamensis* and *P. filipes*). In contrast, species with low λ s have a low ability to recover between harvests and therefore have commercial proportions that drop after each harvest (e.g., *C. spruceanum* and *L. mahuba*). The remaining species with λ s slightly above 1 stabilized at pre-harvest commercial proportions of approximately 0.6-0.7. Under the same scenario, second harvest volume falls to 39 m³ (a 75% stock recovery rate). The inclusion of volume-based harvest limits generally decreased the rate of decline of commercial proportion in modeled projections.

The most intensive management regimes resulted in the largest first harvests (87 m³ with a MCD of 30 cm DBH, 0.9 harvest intensity, and no harvest volume restrictions) but also resulted in very low future harvest volumes (6.2 m³ per ha at fifth harvest under a 30-yr cutting cycle). The optimal management regimes for the four STM indicators varied in terms of most management criteria except for the common prescription of short 10-year cutting cycles and no volume limits per harvests (Table 4-5). The management regime with highest H₃ AY resulted in slightly more aggressive harvests by specifying a MCD of 40 cm DBH. Optimal management regimes for all STM indicators but highest H₃ AY (the most short-sighted STM indicator) included MCDs of 50 cm DBH or greater.

Most of the top mean AY \cap λ_H management regimes required a MCD of 60 cm DBH and volume harvest limits. These STM regimes showed little indication of yield decreases over time and required little additional recruitment to compensate for population growth decreases. All optimal management regimes for the other three STM indicators showed signs of declining yields and higher compensatory recruitment needs. However, of the three, the management regime with highest mean AY also required the least number of compensatory recruitment. The Brazilian legal scenarios resulted in lower mean AY than any of the optimal STM regimes while still requiring similar numbers of compensatory recruitment, indicating potential resource under-utilization.

For each STM indicator we evaluated the differences in optimum management criteria among species. Given the inter-specific demographic differences, there was nearly no overlap among the three best species-specific management regimes according to the four STM indicators (Table 4-6). The inclusion of species-specific

regimes led to better results compared to general optimal management regimes (Table 4-7). The best species-specific management regimes for highest H₅ AY resulted in a 0.34 m³/yr annualized yield gain. The best species-specific management regimes for highest mean AY resulted in a 0.12 m³/yr annualized yield gain.

STM Sensitivity to Management Criteria

Stand mean AY decreases with longer cutting cycles and the inclusion of harvest volume limits, but peaks at a MCD of 50 cm (Figure 4-5). While other management criteria were not significantly correlated to stand mean AY (harvest intensity, minimum commercial density, and seedtree quality), species-specific analyses yielded more nuanced results. Longer cutting cycles lead to large mean AY reductions in *M. paraensis*, but also to small increases to *V. surinamensis*. Harvest intensity had a small negative impact on mean AY for slow-growing species (*M. paraensis*, *C. spruceanum*, and *L. mahuba*). Higher MCD has a small positive effect on mean AY of *M. paraensis*, but strong effects on all other species. However, this relationship was curvilinear with mean AY peaking around a MCD of 50-60 for most species. High MD decreases the mean AY of the least common species (*C. spruceanum*, *L. mahuba*, and *P. filipes*), with its negative effects increasing with species scarcity. Harvest volume limitations reduce mean AY of the fast-growing species (*C. guianensis*, *V. surinamensis*, and *P. filipes*), but cause the opposite effect in *C. spruceanum*.

Stand λ_H increases with higher MCD and the inclusion of harvest volume limits but also suffers mild decreases with increasing harvest intensity. Species-specific λ_H results were consistent with stand λ_H results above with few exceptions. MD has a positive effect on λ_H for the three least common species, indicating a trade off for these species

considering the opposite effect of MD on mean AY of the same species. Increasing cutting cycles also has a mild positive effect on all species but *M. paraensis*.

Discussion

Demography Differences Between Recently Logged and Recently Undisturbed Estuarine Forests

Results from monitoring of harvesting operations show that demographic effects of selective harvesting were small and likely a result of small damage from non-mechanized operations in relatively vine-free forests. The observed weak harvest effects are in agreement with upland forest studies that show tree fall gaps have small and short-lived demographic effects (Finegan and Camacho 1999, Fredericksen and Mostacedo 2000, Sist and Nguyen-The 2002, Smith and Nichols 2005). Crown illumination evaluations and field observations suggest canopy gaps close quickly after logging in the study area, which agrees with assessments of felling damage in upland forest (Asner et al. 2004, Broadbent et al. 2006).

Although stem diameter growth was the demographic rate most affected by selective logging, and that light availability controls diameter growth in undisturbed stands (Chapter 3), the fact that post-harvest growth effects were not driven by changes in light availability is surprising (Chazdon et al. 1996). Given the small size of logging gaps, full canopy closure could occur long before stand density has recovered, resulting in continued underground competition (Kammesheidt et al. 2001). The unexpected lack of growth response by *C. spruceanum* (a light-demanding species) may be due to relatively small canopy openings that benefited its shade tolerant competitors (Peña-Claros et al. 2008). Logging damage seems an unlikely reason the lack of growth response in *C. spruceanum* because damage assessment in logged plots and recently

logged areas showed minimal damage that affects species indiscriminately, hence any damage effect should have affected other species as well.

While other studies have found increased post-harvest mortality persisting years after initial harvests (Schulze and Zweede 2006; Figueira et al. 2008; *but see* Silva et al. 1995), mortality rates in logged plots did not differ from those observed in recently unlogged plots in this study. The lack of mortality effect could be partially explained by the observed low residual stand damage, and relatively small gaps that close quickly. Another possibility may be that, unlike unlogged mature stands elsewhere that may contain many structurally weak trees sheltered by a closed canopy (Gardiner et al. 1997), the long history of logging may have already culled large structurally-compromised trees in previous harvests.

Widespread advanced regeneration of *Mora paraensis* produced a strong increase in recruitment into the smallest measured size class (>5 cm DBH) after logging for that species alone. Because logged plots varied from 1 to 6 years since last extraction, it is possible that the post-harvest recruits of other species had not yet reached measurement size. Nevertheless field observations also show there were nearly no *C. spruceanum* saplings present in logged and natural gaps, indicating the common extent of logging disturbance is likely insufficient for the recruitment of this and other light-demanding species. Low intensity harvests have been found elsewhere to be insufficient to boost regeneration of light-demanding species (Wittmann and Junk 2003). These results indicate that the challenge to promote regeneration of light-demanding species and to simultaneously reduce logging damage extends to the Amazonian

floodplain (Fredericksen and Mostacedo 2000, Fredericksen and Putz 2003, Zarin et al. 2007).

Using *M. paraensis* (the most abundant species) as an example, the integration of demographic effects of logging using life table response experiments (LTRE) shows that demographic rates most affected by logging may not be the most important for determining post-harvest tree population dynamics. These conclusions are important because harvest evaluations commonly consider growth, survival, and recruitment effects separately, and may misrepresent changes in population dynamics resulting from observed demographic effects (Fortini et al. 2010). Without the use of an integrated population approach, however, it would not be possible to detect the larger contribution that smaller juvenile growth increases offer to the persistence of the species in the stand.

Harvest Damage from Non-mechanized Small-scale Logging

Residual stand damage was relatively low compared to other studies (Johns et al. 1996, Pinard and Putz 1996, Jackson et al. 2002, Rockwell et al. 2007b), with no observed damage to large trees during yarding operations. These results bode well for future harvests as these larger trees will likely constitute the second harvest cohorts. This low residual stand damage is likely a consequence of several favorable factors including the low abundance of vines and lianas that hinder directional felling and cause multiple tree falls, the large abundance of palms which may provide 'safe' felling zones (Chapter 2), the lack of heavy machinery use, and the absence of road construction. A proportion of residual stand mortality resulted from the need for transport rails and float wood used in the manual transport of timber from the forest, causes of harvest mortality

not observed in upland logging operations. These practices avoid use of timber species and instead use species of little to no economic value (e.g., *Inga spp* and palms).

Prospects for Sustainable Timber Yield

Most evaluations of stock recovery rates in tropical forests are below 50% over government-specified cutting cycles, casting doubt over the feasibility of long-term STM (Dauber et al. 2005, Brienen and Zuidema 2007, Sist and Ferreira 2007, Schulze et al. 2008, Rondon et al. 2009). In contrast, while the potential for SFM in Amazonian estuarine floodplains has long been recognized, it has been little explored (Barros and Uhl 1995, Pinedo-Vasquez et al. 2001). The fact that timber is still extracted today after hundreds of years of low intensity logging in the estuary supports results from this study that suggest sustaining future harvests is possible.

In this study, recovery rates are high for most simulated management regimes due to the comparatively high projected mean AYs (Van Gardingen et al. 2006) that are close to the maximum estimated tropical forest productivity limits (Rice et al. 2001). These large mean AYs are primarily due to projected increases in yields from the two fast-growing species *V. surinamensis* and *C. guianensis* that effectively counterbalance the projected yield declines from other species. Given the historically high volumes of *C. guianensis* and *V. surinamensis* extracted in the past, the drastic projected recovery of these two species may be explained by the stark differences between common extractive practices and simulated forested management. As described elsewhere, smallholders often practice re-entry logging where tree populations are harvested down to small tree diameters (Barros and Uhl 1995, Lima et al. 2006). In fact, model projections show that the harvesting of all commercial individuals down to small

diameters will lead to rapid population and yield declines for both *C. guianensis* and *V. surinamensis*.

While our results point to increases in the proportion of non-commercial trees due to consecutive harvests, model results suggest that fast population growth and related fast diameter growth preclude the possibility of forests of un-merchantable trees. Nevertheless slow-growing populations exhibit slow recovery in tree densities and volumes after logging and face continuously decreasing proportion of commercial trees while under a timber harvest regime. For instance, while most study species under simulated management reached a new lower proportion of commercial trees under intensive harvests, the two species that showed signs of population decline, *C. spruceanum* and *L. mahuba*, not only exhibited limitations in population recovery but had continuously declining proportions of commercial trees after each harvest cycle (Chapter 3). Fortunately, the number of recruits needed to maintain population stability was low. This need for additional recruits is particularly relevant to the Amazon estuary because, at the smallholder scale of management, additional costs related to managing timber species regeneration are low due to low opportunity costs (Keefe 2008). Local initiatives to improve seedling supply are underway, but a better understanding of enrichment planning may be needed to ensure success (Schulze 2008, Keefe et al. 2009).

Preliminary model results based on applying density dependence using stand level carrying capacity show that forest composition can change considerably over the course of multiple harvest rotations. While few studies explore the consequences of logging on tropical forest composition, those that address it generally conclude these

changes can affect long-term ecological and economic viability of yields (Alder and Silva 2000, Phillips et al. 2004). Nonetheless, prospects for STM are better in the studied stands than for many other areas evaluated elsewhere in the Amazon (Dauber et al. 2005, Rondon et al. 2009).

Lastly, *Callycophyllum spruceanum*'s lack of post-harvest recruitment, despite its ample regeneration in large forest clearings and secondary forests (De Jong 2001, Sears 2003) and commonly observed cohort-like size distributions (Chapter 3) indicates this species is a long-lived canopy pioneer species heavily dependent on large scale disturbances (Condit et al. 1998). The prospects for STM of this species would likely be best evaluated by considering how population dynamics of the species interacts with landscape dynamics of the region. This landscape analysis could be done by integrating a landscape model than considers the probability of disturbances large enough for pioneer recruitment along with appropriate demographic modeling of resulting cohorts.

Determining Best Management Regimes for Sustainable Timber Management

The variability and complexity of population responses to logging observed here and elsewhere suggest limited prospects for simple and effective one-size-fits-all management prescriptions (Dauber et al. 2005, Sebbenn et al. 2008, Schongart 2008). Due to contrasting species demography, the importance of management criteria on yield and population growth varied widely among species. In some cases even universally important management criteria had contrasting effects across species (e.g., harvest volume limits). This complexity of species-specific responses tended to be averaged out in stand-level analyses. For instance, analysis with all species combined showed minimum density limits barely affected yield and population growth, as noted elsewhere (Schulze et al. 2008); However, the same analysis at a species level show

this variable is important in terms of yield and population growth for the few rare species included in the analysis. Surprisingly, the comparison of species-specific vs. general (but study-specific) optimal management regimes did not show substantial differences in yields or population growth under simulated long-term forest management. However, it is likely these results are highly dependent on the range of demographic variability included and the set of species studied. Applying a similar analytical approach to a larger set of species could reveal patterns in optimal management criteria related to commonalities in species demography, thus resulting in demography-specific management guidelines.

Aside from the general and species-specific management guidelines devised in this study, this research shows how population modeling tools can help inform decisions about harvest regulations. Population dynamics-based outputs such as compensatory recruitment, and population growth rates under logging are valuable tools for deciding which management regimes are most appropriate for a given region and species/functional group. Nevertheless, this study shows how optimal STM regimes depend on how STM is quantified. STM indicators with potentially short management horizons, such as the H3 and H5 AY indicators, led to aggressive optimal management regimes that still resulted in large drops in harvest volume between first and later harvests. The other STM indicators, mean AY and mean $AY \cap \lambda_H$, were calculated over a fixed 120 yr management horizon and seem to lead to better STM regimes by providing high AYs while requiring lower numbers of compensatory recruits per harvest. Overall, mean $AY \cap \lambda_H$ seems to be the best performing STM indicator as it prevents decreases in harvest volumes, guarantees high mean AY while requiring the least number of compensatory

recruits per harvest of the four indicators used. However, it gives arbitrary equal weight to yield and population growth and yields results not comparable to other studies since it is a sum of ranks.

While we cannot easily implement different harvest intervals for individual species or species groups due to logistic and economic constraints, we can more readily alter species-specific harvest intensity and MCDs. The importance of MCDs to most species future yield and population growth, and the ease of implementing species-specific MCDs suggest that biologically meaningful and species (or functional group) specific management criteria may provide a feasible combined STM and SFM-oriented strategy (Schulze et al. 2008). A post-hoc analysis for optimal species-specific management regimes under the mean $AY \cap \lambda_H$ indicator using the optimal cutting cycle length and harvest volume limits indicated by the stand analysis results in easily implementable species-specific management regimes (Table 4-5).

Almost all optimal STM regimes exclude the possibility of MCD lower than 50 cm DBH, which bodes well for current Brazilian legislation with similar requirements. A central factor in determining yields of the Brazilian legal management regimes is the inclusion of harvest volume limits. This study shows that such an overarching rule may in some cases lead to resource underutilization. In fact, the mean $AY \cap \lambda_H$ optimal management regimes produces a much higher mean AY with nearly no need for extra compensatory recruitment.

While this study focused on rotation-based management as required by Brazilian law, most smallholders in the Estuary practice re-entry logging that vary from low to very high intensity (Macedo and Anderson 1993, Barros and Uhl 1995). On one hand,

common high-intensity and frequent re-entry harvests have large influences on population persistence as evidenced by the historical decline of populations of *C. guianensis* and *V. surinamensis* (Chapter 3). On the other hand, as evidenced in few smallholder properties in Mazagão, at low harvest intensities it is not clear re-entry logging is more damaging per volume extracted given the generally low harvest damage. On the contrary, field observations and local knowledge suggest that fewer intense harvests may release the undesirable *Astrocaryum* palm in the understory leading to decreased regeneration of woody species. While rotation and area-based management is the current paradigm of tropical forest management, it may drive smallholders in the estuary into illegality partly due to management plans that require long time intervals with no revenue and complex requirements that require costly technical assistance (Hirakuri 2003). Although current Brazilian law specifies simpler timber licensing procedures for smaller properties, future research should explore potential management requirements that are more compatible with small-scale low intensity forest management operations (Hirakuri 2003, Zarin et al. 2007).

Table 4-1. Study species

Family	Species	Local name	Characteristics
Caesalpiniaceae	<i>Mora paraensis</i>	Pracuúba	Recently harvested, shade tolerant, high density
Chrysobalanaceae	<i>Licania heteromorpha</i>	Macucu	Not yet harvested, shade tolerant
Rubiaceae	<i>Callycophyllum spruceanum</i>	Pau mulato	Harvested since 1990s, shade intolerant, pioneer
Meliaceae	<i>Carapa guianensis</i>	Andiroba	Long used, shade tolerant
Myristicaceae	<i>Virola surinamensis</i>	Virola	Long used, low wood density, shade tolerant
Lauraceae	<i>Licaria mahuba</i>	Maúba	Long used, shade tolerant
Sapotaceae	<i>Pouteria sagotiana</i>	Maçaranduba	Low local use, shade tolerant, high wood density
Fabaceae	<i>Platymiscium filipes</i>	Macacaúba	Long used, shade intolerant, high wood density

Table 4-2. Transition matrix model used for management simulations

	C					NC				
	C ₁	C ₂	C ₃	C ₄	C ₅	NC ₁	NC ₂	NC ₃	NC ₄	NC ₅
C ₁	S ₁		F ₃	F ₄	F ₅			F ₃	F ₄	F ₅
C ₂	G ₁	S ₂								
C ₃		G ₂ *(1-x ₂)	S ₃							
C ₄			G ₃ *(1-x ₃)	S ₄						
C ₅				G ₄ *(1-x ₄)	S _{5c}					
NC ₁						S ₁				
NC ₂						G ₁	S ₂			
NC ₃		G ₂ *(x ₂)					G ₂	S ₃		
NC ₄			G ₃ *(x ₃)					G ₃	S ₄	
NC ₅				G ₄ *(x ₄)					G ₄	S ₅

x- Probability that commercial (C) trees growing to next size class will become non-commercial (NC) because of defects. S= survival probability, G= size class upgrowth probability, F= Fertility rate per capita; Numerical subscripts denote 15 cm-wide size classes from 5 cm DBH to maximum observed size

Table 4-3. Values for management criteria used in management simulations

Cutting cycle	Harvest intensity	MDC	Min. com. density	Seed tree quality	Max volume per harvest (m ³ yr ⁻¹ ha ⁻¹)
10*	0.5	30	0	High	-
20	0.6	40	0.03*	Low*	1*
30*	0.7	50*	1		
40	0.8	60			
	0.9*	70			

* denote criteria specified by Brazilian federal management guidelines

Table 4-4. Residual stand damage from monitored timber extraction

BA (m²) of trees >5 cm DBH killed* by BA (m²) of timber extracted

	Tree fall	Transport	Total
Palm	0.17	0.03	0.2
Timber	0.13	0	0.13
Other woody spp.	0.28	0	0.28
Total	0.58	0.04	0.62

BA (m²) of trees >35 cm DBH killed* by BA (m²) of timber extracted

	Tree fall	Transport	Total
Palm	0	0	0
Timber	0.11	0	0.11
Other woody spp.	0.17	0	0.17
Total	0.28	0	0.28

*estimate of mortality also includes trees clearly damaged beyond potential recovery.

Table 4-5. Optimal management regimes defined by alternative sustained timber yield indicators

Management optimized for	Considering	Optimal criteria						Management outcomes			
		Cutting cycle	Harvest intensity	MCD	Min. com. density	Seed tree quality	Max volume per harvest	Volume of first harvest	Volume of fifth harvest	Compensatory recruitment	Mean AY
Largest mean AY	All species	10	0.5	50	0.03	Low	300	28.94	19.34	10	1.8
Largest H ₅ AY	All species	10	0.9	50	1	Low	300	49.91	21.23	12	1.55
Largest H ₃ AY	All species	10	0.5	40	0.03	Low	300	40.35	16.16	20	1.47
High mean AY \cap λ_H	All species	10	0.5	60	1	High	300	15.77	15.02	4	1.44
Brazilian legal scenario	All species	30	0.9	50	0.03	Low	30	30	30	22	1
Brazilian legal scenario	All species	10	0.9	50	0.03	Low	10	10	10	3	1
High mean AY \cap λ_H	<i>M. paraensis</i>	10*	0.5	60	1	Low	300*	18.02	15.91	5	1.52
High mean AY \cap λ_H	<i>C. spruceanum</i>	10*	0.6	60	1	High	300*	18.92	15.66	5	1.48
High mean AY \cap λ_H	<i>C. guianensis</i>	10*	0.8	60	1	Low	300*	28.84	14.50	5	1.48
High mean AY \cap λ_H	<i>V. surinamensis</i>	10*	0.6	50	1	Low	300*	33.27	17.50	10	1.71
High mean AY \cap λ_H	<i>L. mahuba</i>	10*	0.5	50	0.03	Low	300*	28.94	19.34	10	1.8
High mean AY \cap λ_H	<i>P. filipes</i>	10*	0.9	50	1	Low	300*	49.91	21.23	12	1.55

* management criteria fixed based on optimal value from analysis of all species combined

Table 4-6. Comparison of species-specific optimal management regimes according to four STM indicators

	<i>M. paraensis</i>	<i>C. spruceanum</i>	<i>C. guianensis</i>	<i>V. surinamensis</i>	<i>L. mahuba</i>	<i>P. filipes</i>
Largest mean AY						
<i>M. paraensis</i>	1					
<i>C. spruceanum</i>	0	1				
<i>C. guianensis</i>	0	0	1			
<i>V. surinamensis</i>	0	0	1	1		
<i>L. mahuba</i>	0	0	0	0	1	
<i>P. filipes</i>	0	0	0	0	1	1
Largest H ₅ AY						
<i>M. paraensis</i>	1					
<i>C. spruceanum</i>	1	1				
<i>C. guianensis</i>	1	1	1			
<i>V. surinamensis</i>	1	1	1	1		
<i>L. mahuba</i>	0	0	0	0	1	
<i>P. filipes</i>	0	0	0	0	0	1
Largest H ₃ AY						
<i>M. paraensis</i>	1					
<i>C. spruceanum</i>	0	1				
<i>C. guianensis</i>	0	1	1			
<i>V. surinamensis</i>	0	1	1	1		
<i>L. mahuba</i>	0	0	0	0	1	
<i>P. filipes</i>	0	0	0	0	0	1
High mean AY \cap λ H						
<i>M. paraensis</i>	1					
<i>C. spruceanum</i>	0	1				
<i>C. guianensis</i>	1	1	1			
<i>V. surinamensis</i>	0	0	0	1		
<i>L. mahuba</i>	0	0	0	0	1	
<i>P. filipes</i>	0	0	0	1	0	1

Ones denote a match in optimal management prescriptions between species, zeros a mismatch

Table 4-7. Differences in performance between species-specific and general optimal management regimes

	Mean AY ($\text{m}^3 \text{ yr}^{-1} \text{ ha}^{-1}$)	H ₅ AY ($\text{m}^3 \text{ yr}^{-1} \text{ ha}^{-1}$)
<i>M. paraensis</i>	0.03	0.11
<i>C. spruceanum</i>	0.02	0.04
<i>C. guianensis</i>	0.05	0.03
<i>V. surinamensis</i>	0.01	0.01
<i>L. mahuba</i>	0	0.14
<i>P. filipes</i>	0	0
Total	0.12	0.34

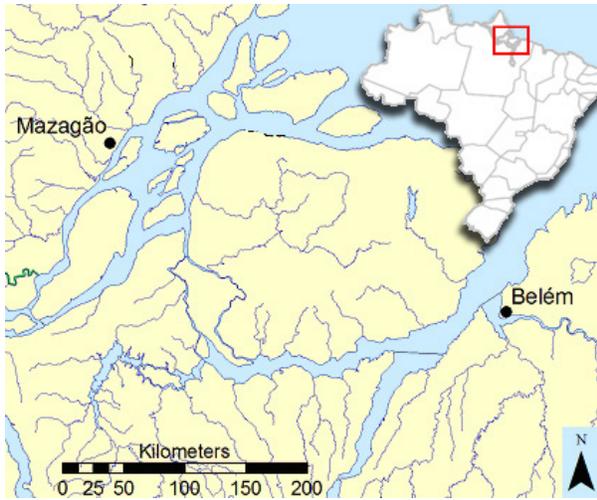


Figure 4-1. Map of study area

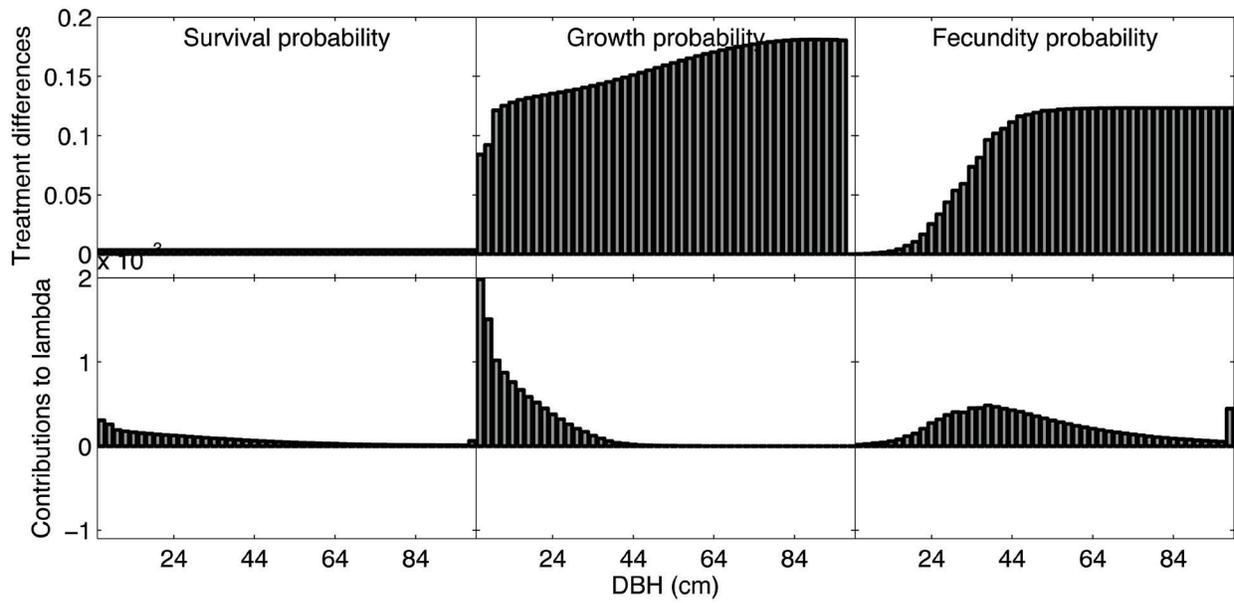


Figure 4-2. Life table response experiment for *M. paraensis* demonstrating demographic differences between logged and recently undisturbed plots and their contributions to population growth

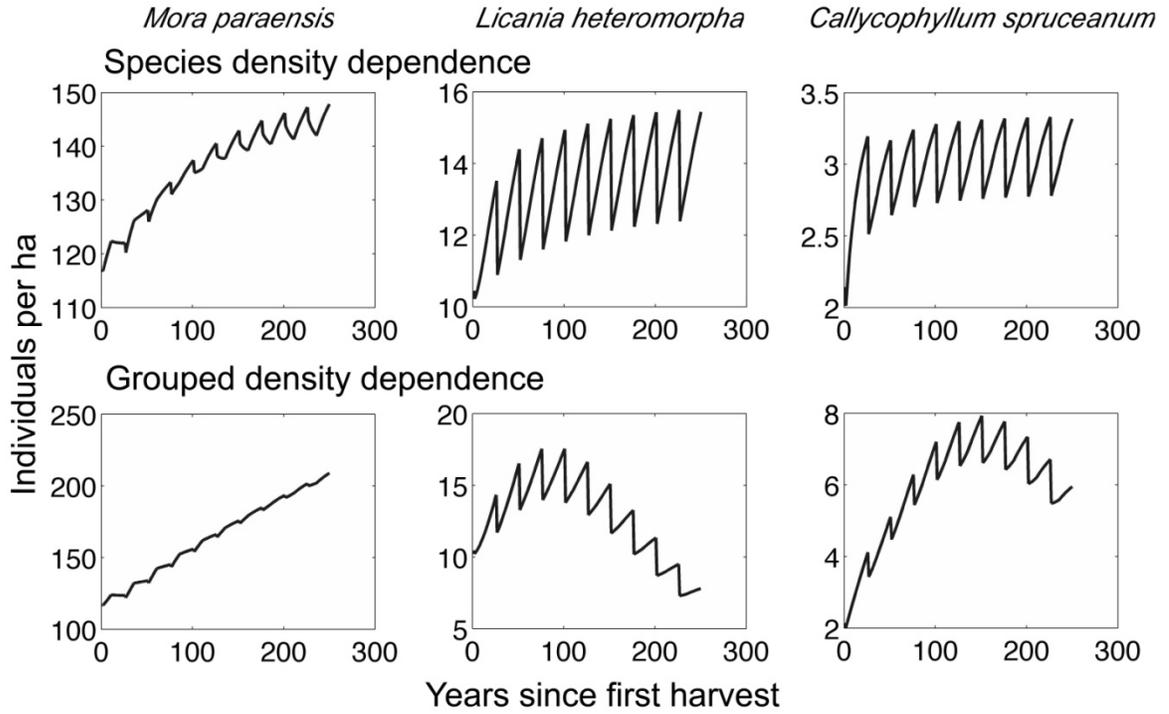


Figure 4-3. Management simulation projections illustrating differences between species and grouped density-dependent models

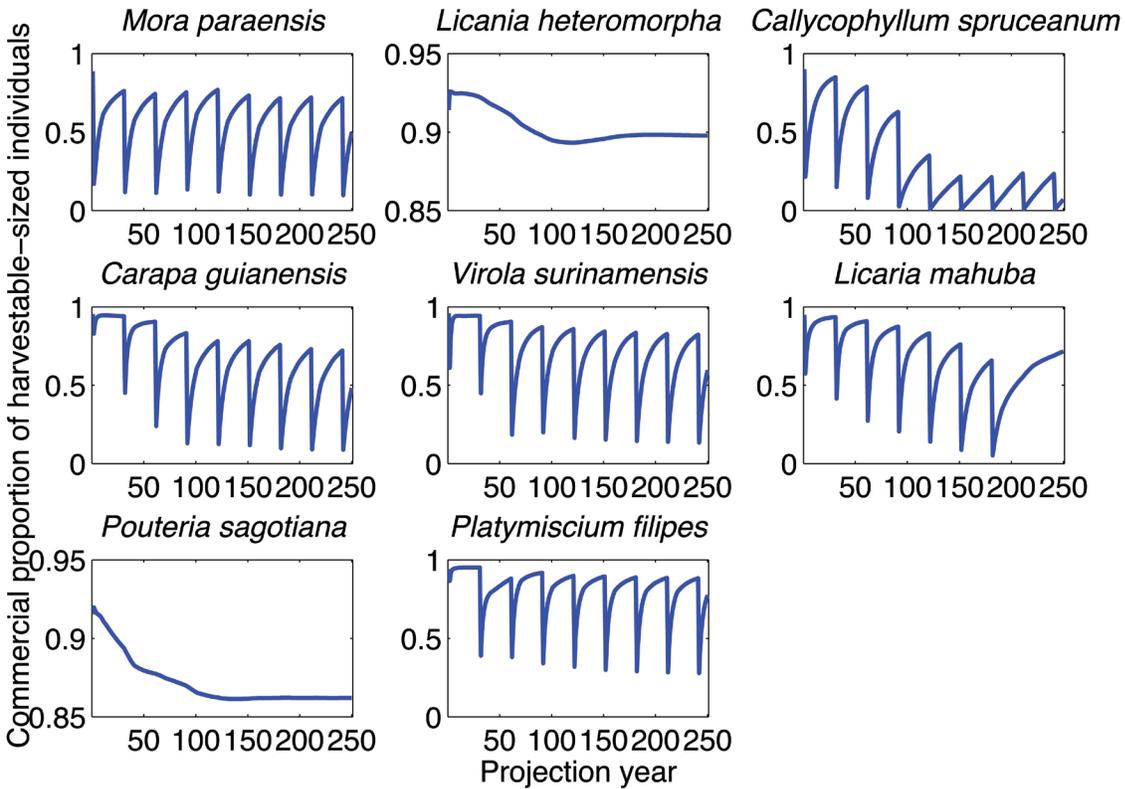


Figure 4-4. Shifts in commercial proportion under the Brazilian legal scenario with 30 yr cutting cycle but without volume-based harvest limits

Licania heteromorpha and *P. sagotiana* were not harvested during simulations. *Licaria mahuba* was harvested until year 180, after which densities were too low for continued harvests.

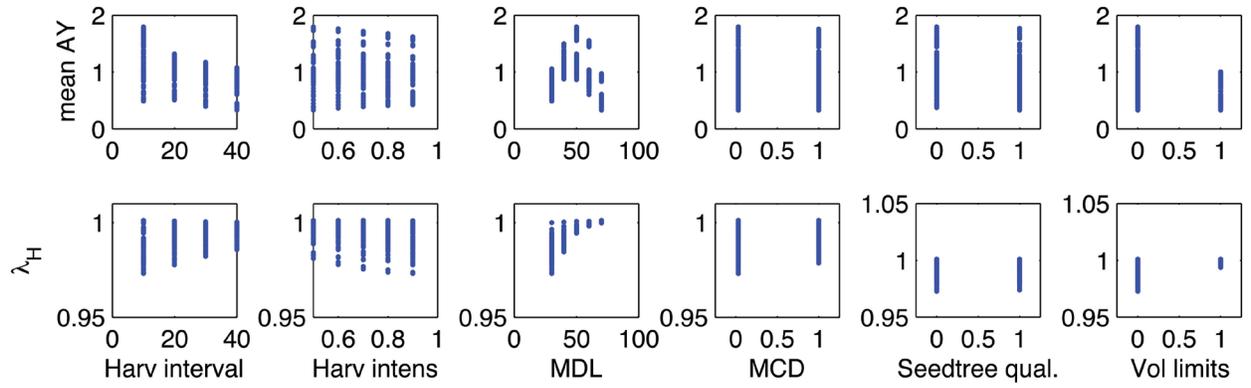


Figure 4-5. Simulation-based spread of mean AY and λ_H in response to varying management criteria.

CHAPTER 5 TIMBER MICRO FIRMS OF THE AMAZON ESTUARY: A VIABLE ECONOMIC MODEL FOR DEVELOPMENT?

Introduction

While the majority of timber management literature from the Amazon has focused on industrial operations, it is estimated that 95% of rural properties in the Amazon are less than 500 ha (IBGE, 1996), providing as much as 28% of regional timber output (Lentini et al. 2005). Government settlements in the Amazon alone account for approximately 500,000 smallholder families that commonly sell timber (Lima et al. 2006). Surprisingly, relatively little attention has been paid to the potentials and limitations of timber management in smallholder scales, with the most relevant research focusing on community forestry efforts (d'Oliveira 2000, Rockwell et al. 2007b).

Smallholder timber operations may vary substantially from industrial operations in techniques, technology, capital availability, market reach and ecological impacts (chapter 4, Salafsky et al. 1998, Rockwell et al. 2007a, Keefe 2008). Without the proper knowledge and consideration of the potentials and limitations of smallholder timber management, most legislation on timber use in the tropics has focused on the industrial scale, leading to unrealistic expectations to smallholders and communities (d'Oliveira 2000, Rockwell et al. 2007a, Zarin et al. 2007).

In the mouth of the Amazon River, smallholders have developed a micro-scale vertically-integrated system of timber production (Pinedo-Vasquez et al. 2001, Sears et al. 2007a). Contrary to community-led efforts elsewhere, these informal micro firms are owned individually and commonly integrate timber extraction and processing in local circular sawmills. For decades, hundreds of these micro firms have produced sawn lumber sold primarily at local and regional markets (Barros and Uhl 1995). As past

research in Amazon tidal forests have documented the potential for small-scale sustainable timber management, these micro firms bear special relevance to the potential role of smallholders in sustainable timber management (chapter 4, Barros and Uhl 1995, Pinedo-Vasquez et al. 2001). Surprisingly, one of the largest threats to micro-firm timber production in the estuary appears to be the açai palm (*Euterpe oleracea*) fruit. Characterized as a non-timber forest products by some or as a pressure for forest conversion by others, it is nevertheless an increasingly popular alternative to timber production in the Amazon Estuary (Brondizio 2004, Weinstein and Moegenburg 2005). Intensified açai production often results in mono-specific stands where most competitors species (including timber) are eliminated while the abundance of açai is increased from natural regeneration and additional plantings.

As one of few examples of small scale localized timber production systems, the Amazon Estuary is ideal for exploring issues of smallholder vertically-integrated timber use. In this study we use data from multiple sources including landowner and firm surveys, participatory monitoring of firms, and detailed forest and sawmill operation monitoring to devise a financial returns model of smallholder timber micro firms and a simpler model for smallholder açai production. We then explore the economics of timber micro firms to address the following questions:

- (1) What are the financial costs and revenues of timber micro firms?
- (2) What micro economic factors most influence long-term economic viability of timber production by micro firms?
- (3) How does timber micro firm profitability compare to regional economic alternatives? How can this be used to advance tropical conservation and development?

Methods

Study Region

We conducted our research in the 160 sq km Mazagão watershed at the Western side of the Amazon estuary (Figure 5-1). The Mazagão watershed has a long history of timber use (Pinedo-Vasquez et al. 2001) with current micro scale timber extraction as part of diverse livelihood strategies that often also include palm fruit, fishing, and cropping. Mazagão is similar in composition and land use history to several adjacent watersheds, as confirmed by region wide inventories and surveys conducted in 2005 (Fortini, *unpublished data*). Mean annual temperature is 27C and average daily temperature varies by less than 3C from month to month. Mean annual precipitation is 2550mm and occurs mostly in the wet season months of January-May. This part of the Amazon estuary is characterized by freshwater tidal fluctuations of 2-3m. Because of the elevated river level in the wet season, local forests may flood twice daily during high tides.

The Várzea Smallholder Timber Production System

The micro scale timber-producing firms are centered around the small sawmills that process locally harvested timber to be sold regionally. Although past research suggests sawmills worked independently from those who extracted timber and sold as logs (Barros and Uhl 1995), forest extraction is increasingly performed by the same 4 – 5 person crew responsible for sawmill operations (Lentini et al. 2005). While up to 10-20 years ago felling by axe was common in the region, now chainsaws are prevalent (Barros and Uhl 1995, Lima et al. 2006). One of the characteristics of these firms is the high dependence in non-mechanized labor. Once trees are felled, tracks are manually cleared where bucked logs are pushed over small rails made from small non-

commercial stems to river edge. Logs are then manually floated with the aid of tides to the mill using float wood or larger rafts (Barros and Uhl 1995). As of 2008, 12 micro firms were installed in the 160 sq km Mazagão watershed.

Monitoring of Extraction and Sawmill Activities

We monitored extraction activities between June and August 2008 to quantify all costs and production of related activities. Because crews alternate time spent in the forest and sawmill, we monitored the activities of two crews. We devised a monitoring methodology to log time spent by each crew in each of five activities: felling and bucking; clearing path and laying tracks, pushing logs, floating logs, and transportation to and from forest sites. All trees and tracks between felled trees and river edge were mapped and geo-referenced. Using a tree, log and trail numbering system, the production for any particular activity (e.g., volume felled, m trail created) was related to time spent on each category. To calculate harvest efficiency, we measured total stem volume (i.e., from base to crown base) and harvested volume for each felled tree by measuring diameter along the stem.

We conducted a similarly detailed monitoring of two sawmills between May and October of 2007. We recorded the processing time, volume and yield of each log processed and all related labor, fuel and food expenses. Additional cost and production data was obtained through the participatory monitoring of three firms from 2006 to 2008. Using simple accounting books, firm owners registered all log and tree purchases by species, listing the place of origin, species, volume or number of logs, and related sawmill processing costs. To check and complement daily cost and revenue estimates with estimates of capital investment, equipment durability and maintenance costs, we also conducted detailed surveys with all firm owners present in the Mazagão watershed

in July-August 2008. Because this study follows 4 years of local rapport building during other related research, we expected our close monitoring to accurately reflect timber activity in the region.

Financial Returns Model

The field data collected was used to construct a single-firm financial return model that assumes firms operate under short run conditions, no market power and no economies of scale for a period of 30 years. The 30-year time horizon was selected to include the lifespan of a sawmill and at least one full typical harvest rotation (10 to 30 years according to Brazilian forestry legislation). To reflect the way timber is usually produced in the watershed, the model includes three modes of production: firms using timber from own property, firms purchasing standing trees, and firms purchasing felled logs.

We used depreciation as a means to annualize periodical replacement costs of capital assets (i.e., an annual contribution necessary to ensure future replacement costs) to avoid peaks in year-to-year costs based on estimates of equipment durability. We calculated straight-line depreciation based on use or time depending on whether the durability of asset was dependent on usage (e.g., mill engine, chainsaws) or time (e.g., mill housing, wood boats). Operational costs were modeled for all stages of production from felling to milling. Since chainsaws are used for extraction and for longitudinally splitting large logs in the sawmill, we divided chainsaw depreciation and maintenance costs according to their proportional use in extraction and milling. Similarly, we excluded a portion of boat and boat engine depreciation and maintenance costs proportional to the estimated boat use unrelated to timber production. The final model also included changes in extraction practices during the flood season (i.e., shorter pushing distances)

and incorporates temporal trends in parameter values. Annual firm revenue was calculated as firm output (based on forest, transportation and sawmill efficiency), allocated among the four predominant product types and their respective prices. Revenue from the sale of solid wood residues to charcoal producers was also included in the model. The model outputs cost and revenues by year and by m³ and calculates NPV of the firm by discounting all costs and revenues over the 30-year projection period. Due to the limited economic opportunities in the region, an interest rate of 6.79 % was used for discounting based on the average rate on savings from 2006 to 2008.

Model Perturbation and Sensitivity Analyses

Following model creation and parameterization, we performed perturbation analysis to evaluate parameter importance to overall model output using multiple model runs with parameters varying randomly within their observed ranges (Boltz et al. 2001). Since results of the perturbation analysis are based on the observed variability of each model parameter, we consequently relied instead on a simpler elasticity analyses that considered the effect of a 1% reduction in mean parameter value on model output NPV. We used a simple formula to evaluate the elasticity of model output to changes in individual parameters (Eq. 4-1).

$$Ela\ NPV_{parameter\ i} = \frac{\frac{\Delta NPV}{|NPV|}}{0.01} \quad 5-1$$

This elasticity formula allowed me to evaluate the direction and percent change in model NPV given a percent change in a particular parameter value.

Evaluating Açaí Costs and Revenues

To compare the attractiveness of timber production to pervasive açaí fruit production, we used previously published data along with interviews with açaí producers in the watershed to quantify startup costs (e.g., clearing and planting), yearly management and harvesting costs (Hiraoka 1992, Munizmiret et al. 1996). Average per ha production estimates for the region were calculated by estimating average açaí stand density from 25 açaí stand inventories and using a second degree, no-intercept polynomial function relating per hectare production and açaí density calculated from previously published data (Brondizio and Siqueira 1997).

$$Production(baskets) = 3.56 \times 10^{-4} \times a\acute{c}a\acute{i} \ clumps^2 + 0.243 \times a\acute{c}a\acute{i} \ clumps \quad 5-2$$

Since production is highly seasonal and largely synchronous across households, we used production and price data collected from a single household between 2005-2008 to calculate weighted average revenue per 18 kg basket of açaí for the region (Munizmiret et al. 1996). Startup costs, annual revenues and management costs were used to calculate the NPV of establishing a 1 ha stand of açaí from recently undisturbed forest and managing it for 30 years. Unlike economic analyses performed elsewhere (Munizmiret et al. 1996), açaí stand NPV was calculated as a finite payment series discounted from year of first harvest minus initial startup costs (Klemperer 1996).

$$NPV = p \times \left[\frac{1 - (1+r)^{-26}}{r} \right] \times (1+r)^{-3} - C_0 \quad 5-3$$

Where p is annual net revenue of producing years, r is discount rate, and C_0 is startup costs. The first half of the equation gives the discounted value at year 3 of p from years 4-30 (i.e., 26 harvest years). The resulting product is then discounted from third year dollar values to the present. This approach was preferred because local açaí

management includes the continuous thinning of older less productive stems that allows for continuous production without the need for the clearance and replanting of the entire stand every 15-20 years.

Results

Extraction Practices

The annual costs of extracting 750 m³ of standing volume to sawmill harbor (the average per firm in the watershed) was \$4457. This value excludes stumpage costs and results in 532 m³ of timber reaching the sawmill based on a 71% timber extraction efficiency. Costs per extraction activity standardized by volume output show floating costs to transport timber from forest river edge to sawmill is larger than all other activities including felling and bucking (Figure 5-2). As expected due to the low level of mechanization, labor-related expenses (6.62 \$/m³) entirely overshadow fuel and equipment costs (0.49 \$/m³ and 1.27 \$/m³, respectively; Figure 5-3). Forest monitoring also revealed that on average 15% of labor time is spent in travelling to and from the forest.

Sawmill Practices

Based on detailed sawmill monitoring data, firms produce on average 195 m³ of sawn wood per year with a mean sawmill processing efficiency from round wood to sawtimber of 37%. A two-way anova also revealed differences in milling efficiency between sawmills and among species sawn (Species df = 4, F = 3.36, p < 0.01; Mills df = 1, F = 12.83, p < 0.0004; Species*Mills df = 3, F = 4.84, p < 0.002). While mean processing efficiency between sawmills varied little (0.41 vs 0.36), processing efficiency varied widely among species, with *M. paraensis* and *P. filipes* having mean values around 0.3 and *C. spruceanum*, *C. guanensis* and *L. mahuba* having mean values

around 0.4. Surprisingly, while volume input and output per log were strongly related ($r = 0.82$, $p < 0.0001$), log use efficiency was not correlated with log diameter. This is likely a result from an increased proportion of irregularities and defects with age and due to the use of chainsaws to split large logs that sawmill saws could not process. Perhaps due to these patterns, while it clearly took longer time to process larger logs ($r = 0.61$, $p < 0.0001$), volume output per hr was not related to log size. Lastly, a link between volume processed daily and person hours of effort was apparent (Figure 5-4; $r = 0.75$, $p < 0.0001$).

Cost and Revenues of Micro Timber Firms

Firms need approximately \$7451 to cover startup costs (e.g., sawmill machinery and housing, chainsaws, assorted equipment) and spend considerably more on milling than in forest operations (47.35 vs 8.38 \$/m³ output, respectively). Only when purchasing extracted timber as logs does the cost of raw materials approach yearly processing costs (\$6892 vs \$9246, respectively; Figure 5-5). On the other hand, average yearly depreciation and maintenance costs (619 and 968 \$/yr, respectively) are relatively low due to the highly labor-dependent system of production. Gross revenue per m³ of sawn lumber produced was \$95.75. In comparison, average cost per m³ produced for firms using their own timber and purchasing standing timber were \$70.19 and \$77.68, respectively. For firms purchasing logs, average cost per m³ produced was slightly below gross revenue per m³, \$89.83. According to model simulations, all modes of firm production were very profitable (Table 5-1); However, differences in profitability among the three modes of firm production were large and resulted in differences in the time required to recoup initial startup costs and internal rates of return (IRR; Table 5-1).

Model elasticity analyses revealed labor-related parameters such as daily wages and work hours per day exert important influence over the NPV of a micro timber-producing firm (Table 5-2). Discount rate also has a very large influence over a firm's NPV, especially because it may vary largely above the range of values used in calculating elasticity. Given the high costs of processing, several sawmill production parameters had a large influence over NPV as well (e.g., mill processing efficiency, processing capacity and days of operation). Asset-related values had minimal impacts on the profitability of a firm, whether in terms of initial price, durability, salvage value or maintenance. Surprisingly, output of extraction activities had a generally small impact on NPV, with only output for floating logs having a moderate effect over NPV.

Açaí Fruit Production Costs and Revenues

Average startup costs for clearing and planting 1 ha of açaí was \$948. Yearly revenue based on the seasonality of production and average açaí stand density in the region is \$1040. Yearly management and harvesting costs are approximately \$518. Nearly all costs associated with açaí fruit production were labor-related since no special equipment is needed for the activity (except for a \$41 yearly expense in harvest baskets). NPV for the establishment of 1 ha of açaí was \$4222.

Timber and Açaí Comparison

NPV over initial investments show timber production yields a better return per dollar invested than açaí fruit production only when firms mill their own timber or timber purchased standing (Table 5-1). Payback periods to recoup timber production startup costs varied greatly depending on the source of raw material (2-9 years; Table 5-1). açaí payback period was also long at 8 years, due to the initial 4-year wait between planting and full production. Based on average timber stocking of regional forests

(Chapter 4), to recoup startup costs micro firms would have to harvest at least 22 ha of their own forests or process as much as 95 ha of timber as purchased logs (excluding harvest schedule discounting; Table 5-1). On the other hand, açaí producers can offset the initial startup costs of stand establishment by selling 9 ha worth of standing timber per ha of açaí stand planted.

Discussion

The Smallholder Timber Micro Firm

The total cost of timber extraction and transport of \$11.12 per m³ of this study is among the lowest values reported for tropical forests for either conventional and reduced impact logging (Verissimo et al. 1992, Barreto et al. 1998, Holmes et al. 2002, Pokorny and Steinbrenner 2005). This particularly low cost is due to a combination of logging costs slightly below other studies and transportation costs within the lowest reported elsewhere (\$7.37 and \$3.75 per m³, respectively). Estimated stumpage costs per m³ are also within the lowest bound of reported values (\$2.75/m³) being only larger than those reported by small-scale conventional logging operations (Verissimo et al. 1992, Stone 1998). While there are generally few economic studies of timber use in the Amazonian floodplains (Barros and Uhl 1995, Lentini et al. 2005), this study supports the notion that varzea extraction can be substantially cheaper than upland timber extraction (Barros and Uhl 1995).

Despite the high incidence of buttressing of floodplain trees (Parolin et al. 2004), timber use efficiency of monitored forest operations was high. This result could be partially explained by the small scale of extraction which led to no observed lost logs or felled trees and the utilization of logs as small as 20 cm in diameter, resulting in little stem volume unutilized below the crown. Additionally, the apparent selectiveness of the

sawyer (who commonly is also the firm owner who pays landowners per tree harvested) means partially defective trees were avoided. Vertical integration of production means micro firms buying or harvesting their own standing trees can select trees that give better return with no incentive to intensify harvests per area. On the other hand, since some firms pay per trees extracted, trees that fall accidentally during harvests and that otherwise would be partially utilized may be left in the forest if it is deemed not worth the tree-based stumpage price.

The estimate of processing costs per m^3 is in par with average Amazon-wide estimates, when corrected for recent exchange rate fluctuations (\$47 vs. \$51 per m^3 , respectively; Lentini et al. 2005). However, this study's processing cost estimate is notably higher than previous survey-based exchange-rate-corrected values for similar circular sawmills (\$47 vs. \$26 per m^3 , respectively; Lentini 2005). The Estimate of sawmill processing efficiency obtained from detailed per log monitoring (0.37) is higher than those published for similar micro sawmills elsewhere (0.28 and 0.35; Barros and Uhl 1995, Lentini et al. 2005a, respectively) but within range of Amazonian-wide industry estimates (Lentini et al. 2005). On the other hand, the present estimates of yearly processing volume per firm are markedly lower than previous survey-based studies. Based on measured average sawmill daily processing capacity and mean number of days of mill operation per month, the 532 m^3 of timber processed annually by sawmills in this study is only a third of values published in previous studies (Barros and Uhl 1995, Lentini et al. 2005). This difference is likely partially explained by the fact that nearly all timber in the present study was purchased standing (and not as logs, thus requiring all firms labor split between forest and mill operations) and smaller average

sawmill crew size. Nevertheless it is unclear how methodological differences may have also shaped these differences, as this study relied on long-term rapport with fewer firms and detailed quantitative monitoring while previous studies utilized survey methods of a much larger number of firms (Barros and Uhl 1995, Lentini et al. 2005).

The micro firms dependence on manual labor instead of oil-subsidized mechanized work means costs of pushing and floating logs are relatively high compared to other harvesting costs and explains why low density (i.e., lighter, more buoyant) timbers have been traditionally preferred. Although the current analysis is based on the processing of a mix of timber species with varying wood density, the observed differences in handling and processing difficulty between high and low density species suggest that a shift in harvests towards either end of the spectrum would likely impact production costs. This link between wood density and labor costs is particularly relevant in the Mazagão watershed where previous research has shown that current intensive reentry logging practices may be suppressing long-term yields of low wood density, high-value species (*Virola surinamensis* and *Carapa guianensis*; Chapter 4).

Micro Firm Profitability

In general, timber production by estuarine micro firms is extremely profitable. This extremely high profitability is likely a consequence of multiple favorable factors. Besides the low stumpage, extraction, transportation and milling costs and no legal costs, as vertically integrated operations, micro firms profit from both extraction and processing. Although IRRs are rarely reported elsewhere, high profitability may be common in the Amazon timber industry, with legal RIL timber extraction yielding an IRR of 35% (Bacha and Rodriguez 2007). Lastly, as Mazagão smallholders have an average property size of 27 ha (Pinedo-Vasquez et al. 2001), most firms likely rely heavily on

timber purchased from other properties in the watershed, thus making the prospects of long-term high profits from harvesting own forests unlikely.

Economic model elasticity analysis yielded an unprecedented look at the relative importance of production factors to firm profitability. As a labor dependent production system, labor related factors were very important but an area where little could be done to improve profitability (e.g., the current daily wage rate is already low and work days are long). Due to the contribution of processing costs to total firm costs, low quality of mill output and the importance of product price to total revenue, improvements in sawmill processing are likely to have a long-term impact on the economic viability of these micro firms. Yet, it is still remarkable that overall costs per unit volume are so low and the activity remains profitable given the low level of technology employed in this production system.

Timber vs Açaí Production in the Amazon Estuary

The choice between investing in timber or açaí for a smallholder in the Amazon Estuary is complex and yields drastically different outcomes. While timber is largely more profitable to micro firm owners than açaí, net yearly revenue per firm is small with a system of production that is not easily scalable. A firm's typical small size is likely what has allowed the industry to operate largely in the informal sector. Micro firms also require much larger initial investments than açaí production. Initial açaí investment can be much smaller and is easily scalable by the size of area to be planted. However, açaí has a delay between planting and production commonly between 3-5 years (Hiraoka 1995). While cropping (e.g., manioc, bananas) in the Amazon estuary ensured a continuous revenue stream during this waiting period (Hiraoka 1995), cropping in the region has nearly vanished due to competition from cheaper imports (Almeida 1996).

While the two activities provide employment opportunities due to high labor use and low dependence on outside inputs, açai production startup and maintenance costs are almost entirely based on household labor and not capital, resulting in minimum cash outlays (Anderson and Jardim 1989). However, açai production is highly seasonal (Munizmiret et al. 1996), making the full reliance on the activity challenging if household finances are not carefully managed. In contrast, a timber firm's year-round operations commonly require at least 4-5 workers and thus often require paid wages.

One major disincentive for smallholders to produce timber is the challenge to operate legally. Current harvesting licensing procedures seem incompatible with small scale timber production (Hirakuri 2003, Scherr et al. 2004). While recent laws attempt to address this issue by simplifying small-scale forestry licensing procedures, licensing is still very costly, requiring technical assistance that results in the dependency on outside institutional support (Hirakuri 2003). Consequently, while açai fruit harvesting is highly physically demanding and dangerous due to tree climbing, unregulated timber production involves the performance of even more dangerous tasks under an absolute lack of safety procedures and equipment.

Environmentally, while açai at the intensity of planting observed in the region can be classified as forest conversion (Weinstein and Moegenburg 2005), it is not necessarily the worse alternative to timber extraction since it is an intense land use and a conversion to a forest type that still provides some valuable environmental services (Brockerhoff et al. 2008, Paquette and Messier 2010). For an average firm extracting 532 m³ of timber yearly, operating under the legal extraction limits of 10 m³ every 10 years or 30 m³ every 30 years per ha requires a minimum management area of 532 ha.

On the other hand, only 12.5 to 18.6 ha of permanent açai cultivation is needed to provide the same NPV as a timber firm processing purchased trees or their own trees, respectively. This simple calculations show how the 'açaiization' of the estuary (Hiraoka 1995) has the potential of changing the dynamics of human disturbance in the Amazon estuarine forests from widespread nearly ubiquitous logging disturbance (Chapter 3 and 4) into highly intensified, albeit still forested açai production areas.

Within diversified livelihood systems common in the Amazon Estuary (Anderson and Ioris 1992), it is not surprising to find timber and açai are to some degree complementary and integrated. Firstly, açai and timber are not temporally exclusive activities because açai harvests peak in the dry season when low water levels makes access to distant timber harvest areas difficult. Harvest monitoring also revealed some level of integration between the two activities. Chainsaw operators showed concern for unnecessary damage to wild açai stock and directionally felled trees away from açai clumps. Logging crews commonly felled old less productive wild açai stems to use as rails for pushing logs from forest to stream. Loggers were aware this practice was beneficial to the wild açai stock as thinning of old stems is a common practice to ensure continuous açai production (Anderson and Jardim 1989). While the \$124 timber subsidy per hectare of forest converted to açai is far from sufficient to cover initial planting costs (based on an average of 16 harvestable trees per hectare, sold standing at \$7.77), many locals showed some degree of preference in converting recently logged areas into açai stands.

Micro Scale Timber Production: A Poverty Driven System?

The price of micro firm timber is limited by its low quality and quantity of production that restricts sales mostly to regional markets and principally for low-income

housing (Pinedo-Vasquez et al. 2001). This low price has left local firms in a challenging situation as, with the spread of açai, fewer landowners are interested in selling timber given the currently low stumpage prices. A secondary effect of the spread of açai management is the increase of local wages alleged by local firm owners. With açai producers during the harvest season easily earning 2-3 times the daily wage rate by harvesting açai, firm owners now have less bargaining power in negotiating wages with potential employees. While an improvement for the local workforce, wage rate increases have a large influence on the economic viability of timber micro firm operations.

In the preceding analyses, all daily wage costs were calculated using the standard regional rate whether the wage was supplied by the entrepreneur's household or not. To a timber or açai entrepreneur, however, labor provided by household members often does not require cash payments and is likely valued much lower than daily wage rates as limited investment alternatives and few employment options lower the opportunity cost of household labor (López-Feldman and Edward Taylor 2009). By alternatively computing NPV of cashflows to and from entrepreneur's household (i.e., setting household labor rate as zero and assuming two household workers available based on local observations), the relative financial attractiveness of the two activities (in terms of NPV per initial investment) changes significantly (Table 5-3). While these values show that at the household level açai management becomes much more worthwhile, these values are only valid within a limited size of açai management area that can be managed by household members. These results may partially explain why açai management is being increasingly adopted in the Amazon estuary generally at

scales smaller than 10 ha per household while the number of micro firms in the region have been on a long decline (Barros and Uhl 1995, Hiraoka 1995, Lentini et al. 2005).

While this research explores the economic rationale and consequences of Amazon Estuary smallholder timber and açai management, questions regarding which of these two activities are more compatible with conservation goals are left unanswered. As both activities currently provide positive returns to smallholder investment, their comparative effects on landscape-level carbon balance, erosion, plant, fish and faunal populations should be explored. If açai is deemed a better choice in balancing conservation and development goals, a program targeted at those currently pursuing timber related activities could provide loans for açai planting to help overcome startup and management costs in pre-production years. If timber is deemed the better alternative, research leading to improvements in sawmill processing technology could have large impacts in the profitability of the activity by cutting costs and improving product quality, along with help in reaching farther markets. Additionally, as most micro firms operate informally, assistance in legalization of the sector would likely have large impacts on timber production in the region.

Table 5-1. Economic indicators of timber and açai management

Activity	Production mode	NPV	Initial investment	IRR	NPV/ Investment ratio	Payback period	Ha of logged forest to recoup startup costs	Equivalent annual annuity
Timber	Purchased trees	\$40,296	\$7,451	84.00%	5.41	3	31.2	\$3,179.11
Timber	Own timber	\$60,105	\$7,451	202.00%	8.07	2	22.1	\$4,741.87
Timber	Purchased logs	\$8,206	\$7,451	52.00%	1.10	9	95.4	\$647.39
Açaí		\$3,230	\$1,114	22.00%	2.90	8	9.0	\$254.82

Table 5-2. Model parameters and their elasticities

Description	Mean estimate	Units	Own timber	Purch. trees	Purch. logs
Discount rate	0.0679		-0.71	-0.75	-1.2
Wages	18.83	R\$ day ⁻¹ person ⁻¹	-0.93	-1.39	-5.07
Work hours per day	9	hrs day ⁻¹	2.08	3.1	11.09
Diesel price	2.22	R\$/L	-0.31	-0.46	-2.26
Gasoline price	2.67	R\$/L	-0.06	-0.09	-0.12
Sawmill diesel consumption	3.26	L/m ³	-0.45	-0.67	-3.29
Forestry efficiency	0.71		0.23	0.84	0.03
Stumpage cost	15	R\$ tree ⁻¹	0	-0.49	0
Extracted timber price	25		0	0	-11.36
Mean volume per tree	3.98	L/m ³	0.08	0.61	0.01
Mean distance to stream	41.39	m	-0.05	-0.07	-0.01
Mill efficiency	0.37		4.06	6.06	29.72
Mean days of mill operation	9.59	d mo ⁻¹	1.47	1.71	3.96
Mill processing capacity	0.14	m ³ person ⁻¹ hr ⁻¹	1.19	1.77	8.68
Mill engine price	4000	U\$	-0.06	-0.09	-0.44
Mill engine use	12.83	yr	0.03	0.04	0.19
Mill engine salvage value	0.31		0.01	0.02	0.08
Mill equipment maintenance	774.33	R\$ yr ⁻¹	-0.09	-0.13	-0.66
Output for felling and bucking	2.64	m ³ person ⁻¹ hr ⁻¹	0.1	0.15	0.02
Output for clearing path and laying tracks	12.21	m person ⁻¹ hr ⁻¹	0.08	0.12	0.01
Output for pushing logs	44.12	m ³ m _{pushed} person ⁻¹ hr ⁻¹	0.17	0.25	0.03
Output for floating logs	0.4	m ³ person ⁻¹ hr ⁻¹	0.45	0.68	1.66
Product A 4m length price	23.1	R\$ dozen ⁻¹	1.54	2.3	11.3

Table 5-3. NPV of household cashflows for timber and açai management

Activity	Production mode	NPV	Initial investment	NPV/ Investment ratio
Timber	Purchased trees	\$96,290	\$7,451	12.92
Timber	Own timber	\$116,099	\$7,451	15.58
Timber	Purchased logs	\$49,807	\$7,451	6.68
Açaí		\$8,898	\$166	53.60

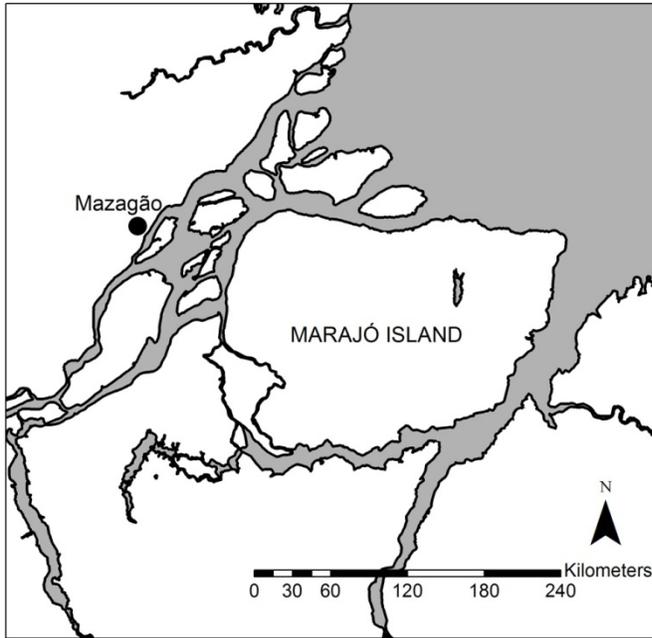


Figure 5-1. Map of study site.

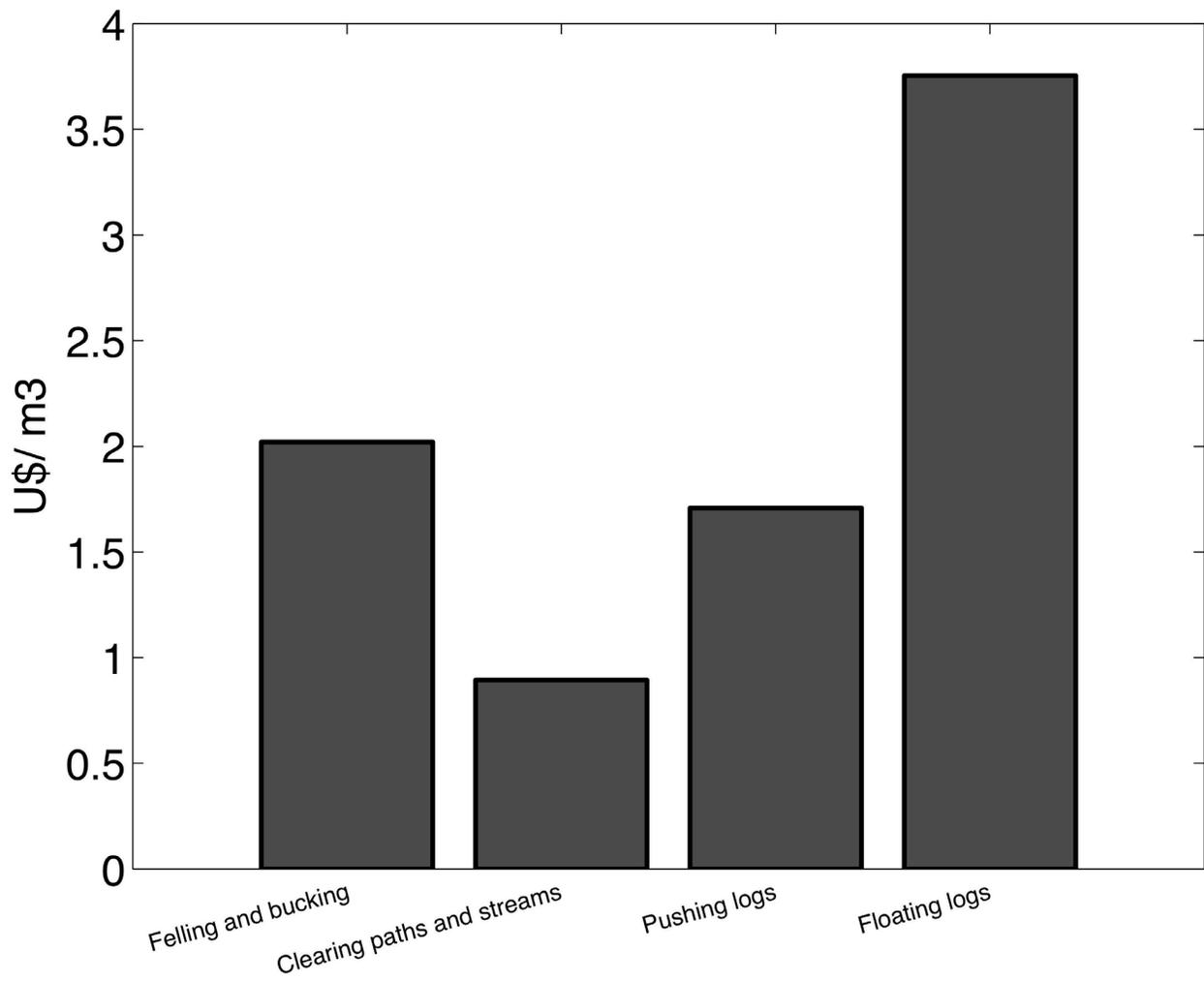


Figure 5-2. Average costs per extraction activity standardized by timber volume delivered to sawmill.

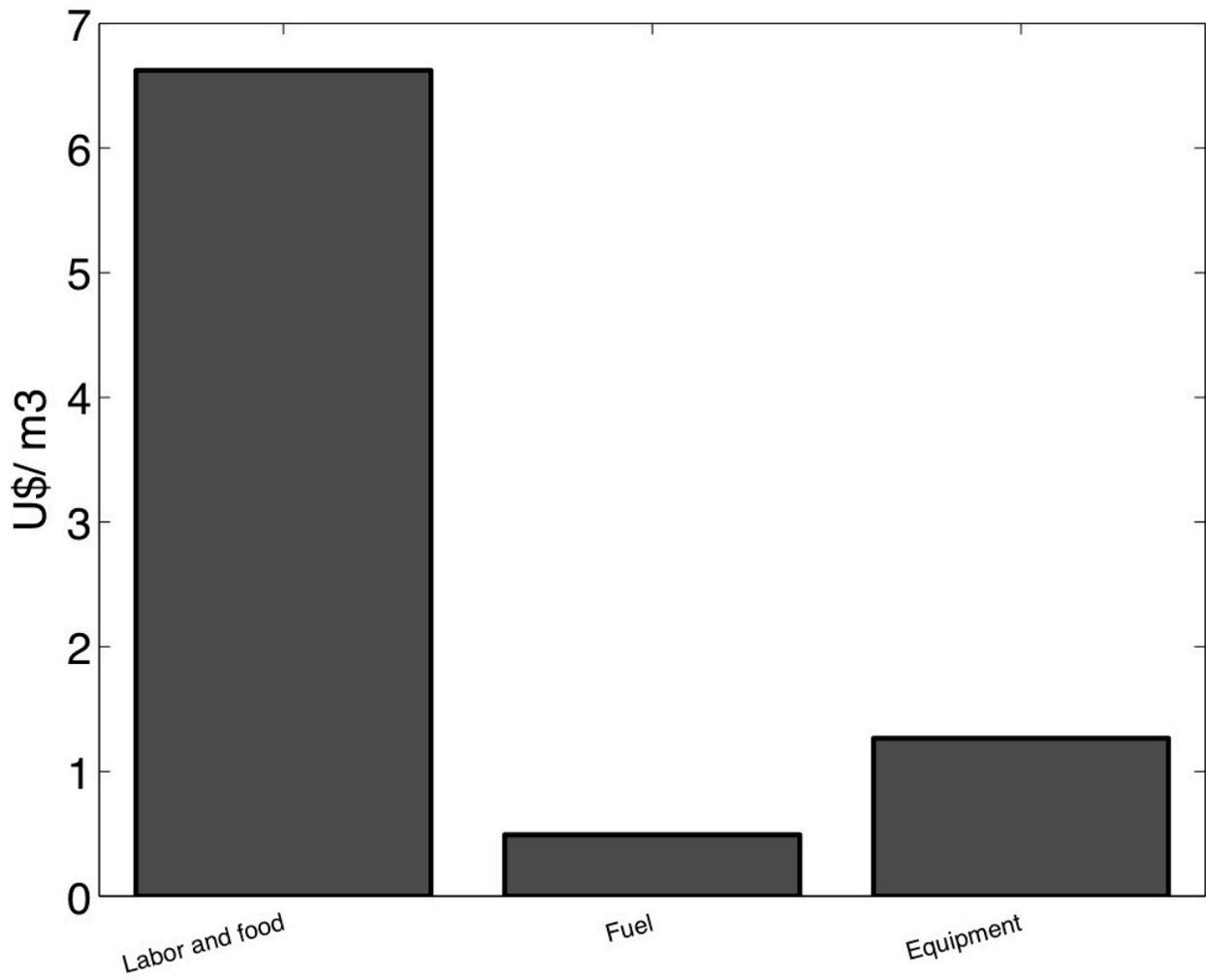


Figure 5-3. Average extraction costs per m³ of timber delivered to sawmill by category

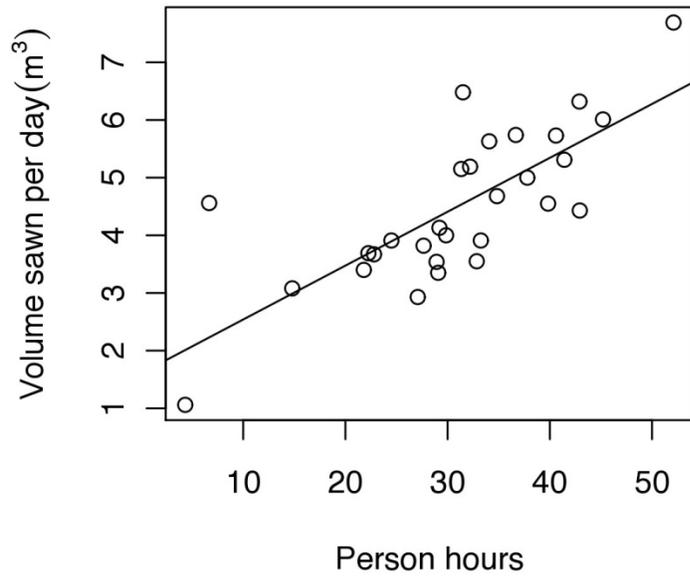


Figure 5-4. Sawmill volume processed daily by person hours of work

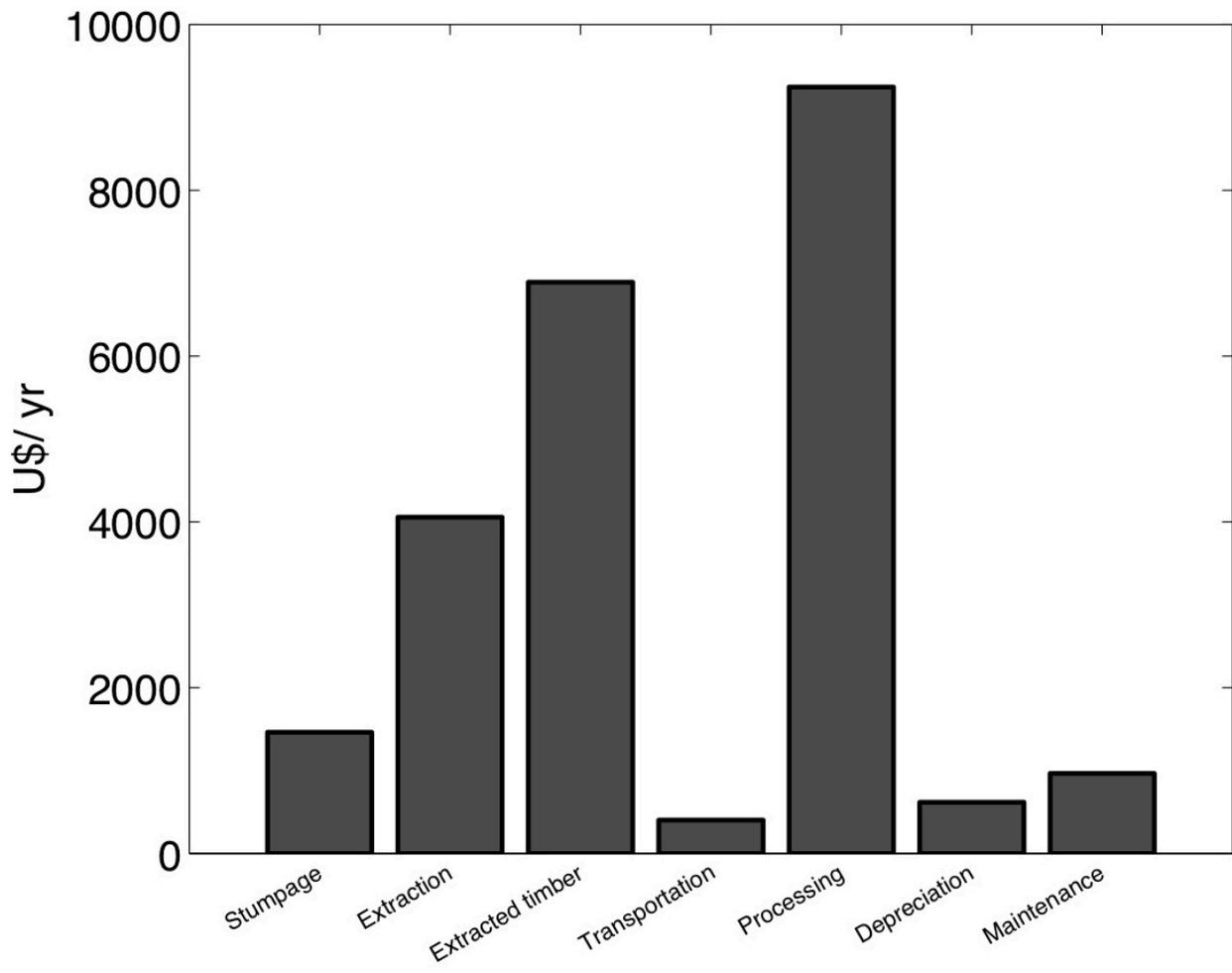


Figure 5-5. Average firm's yearly costs and revenues (processing, forest)

CHAPTER 6
SYNTHESIS: DETERMINING THE ECOLOGICAL AND ECONOMIC VIABILITY OF
TIMBER MANAGEMENT IN THE AMAZON: A WATERSHED SIMULATION
APPROACH

Introduction

Integrated ecological and socio-economic watershed models have been employed to address a variety of complex land use and conservation issues (Beaulieu et al. 1998, Okumu et al. 1999, Gassman et al. 2002, Costanza et al. 2002, Sankhayan et al. 2003, Lant et al. 2005). These models permit analysis of management alternatives and allow for the simulation of complex scenarios that help unravel underlying dynamics, interactions and limitations of study systems (Wear et al. 1996, Weinberg et al. 2002, Costanza et al. 2002, He 2003, Sankhayan et al. 2003, Machado et al. 2003). A watershed modeling approach is particularly well-suited to contribute to the examination of the sustainability of tropical forest use (Rice et al. 1997, Putz et al. 2000, Richards 2000, Pearce et al. 2003) because the models provide explicit focus at spatial scales where the principal ecological and socio-economic factors that constrain sustainability are effectively linked.

In the Amazon Estuary, hundreds of family-run vertically-integrated micro firms harvest, transport and process timber informally to be sold at local and regional markets (Barros and Uhl 1995, Lentini et al. 2005). Consequently, timber remains an important source of income for many smallholder families. At the Mazagão watershed in the Amazon Estuary, following studies of community composition and timber species population ecology (Chapters 2 and 3), sustainable timber management (STM) regimes for local stands have been found (Chapter 4) and the economic viability of micro firms has been explored (Chapter 5). In this chapter we integrate previous ecological,

management and economic models (Chapters 3, 4, and 5) to evaluate watershed timber resource availability and use under alternate scenarios of management, legality, and production levels (Shao et al., 2005). We employed multiple forest use scenarios to address questions of sustainability at the scale of a whole watershed (and its related local timber industry) to explore the link between ecological, economic and legal constraints to forest use and proposed management:

What is likely to happen to the Mazagão watershed if the local industry fails to adopt forest management guidelines?

Do federal management guidelines (FDM) included in Brazilian law improve long-term viability of local industry?

How does a regionally-derived optimal STM management regime perform in terms of ecological and economic viability compared to federal management guidelines or extraction without management?

What are the ecological and economic consequences of legalizing the local industry?

Can micro firms operate to achieve the same return on investments as alternate land uses in the region?

Methods

The Várzea Smallholder Timber Production System

The 160 km² Mazagão watershed has a long history of timber use (Pinedo-Vasquez et al. 2001) with current small-scale timber extraction. Mazagão is similar in composition and land use history to several adjacent watersheds, as confirmed by region-wide inventories and surveys conducted in 2005 (Chapter 3, Fortini et al. 2006).

The micro scale timber-producing firms are centered around small sawmills that process timber harvested in the floodplains of the watershed to be sold regionally (Chapters 4 and 5). Once trees are felled, tracks are manually cleared where bucked logs are pushed over small rails made from small non-commercial stems to river edge.

Logs are then manually floated with the aid of tides to the mill using float wood or larger rafts (Barros and Uhl 1995). As of 2008, 12 micro timber producing firms were installed in the Mazagão watershed.

Mazagão Watershed Simulation Model

The watershed model is based on the integration and scaling up of a demographic population model, a stand management simulation model and a single-firm financial returns model (Chapters 3, 4, and 5).

The underlying demographic model is based on a multi-species equation-based matrix model created for 9 timber species that account for the majority of commercial volume extracted in the region over the past century as well as species that are likely to have timber value in the future (Chapter 3). Based on the demographic model, the stand management model uses a multi-site approach to model size-dependent shifts in commercial and defective portions of each species population (Morris and Doak 2002) and incorporates the effects of density dependence, harvest damage and post-harvest demographic effects in its simulation. The stand management simulation model allows for the specification of several management criteria including harvest rotation length, minimum cut diameter (MCD), intensity (1 - proportion of trees > MCD left as seed trees), seedtree quality (whether or not to include defective trees in seedtree proportion), and minimum commercial density per ha (MD; Chapter 4).

The firm-based financial returns model is a mechanistic model based on the detailed monitoring of extraction and milling activities, participatory monitoring of three firms and detailed surveys with a majority of Mazagão firm owners (Chapter 5).

Scaling and Integrating Precursor Models to Watershed/ Industry Scale

Determining mature forest extent- To scale the stand management model to the Mazagão watershed scale, we used an estimate of available forest area based on the classification of watershed landcover (Figure 6-1). Given the small size of individual palm stands, clearings and secondary forests, we used a combination of high spectral resolution Landsat TM imagery and high spatial resolution Quickbird imagery for the classification. After attempts to classify landcover using multiple statistical pattern-recognition methods (e.g., pixel-based classification algorithms and indices, segmentation, object-based classification methods, and crown recognition approaches) we resorted to heads-up digitizing given the small size of the watershed. Initial data layers used in the digitization included Landsat TM tasseled-cap wetness index (Crist and Cicone 1984) useful for upland/ floodplain forest differentiation, and a water cover map created by a custom Matlab segmentation routine applied to Quickbird imagery. Clearings, shrub-dominated areas, mature, secondary and palm forests were further differentiated using a Gran-smidt pan-sharpened Quickbird image (0.6 m spatial resolution), a principal component analysis image output of Quickbird bands (useful for palm forest differentiation). We also used 338 ground control points collected randomly across the watershed, including nearly two hundred 40 m diameter circular plots to aid the classification. The base Quickbird image was georeferenced with a root mean squared error of 1.2 m, meaning GCPs were likely correctly located on the map.

Model update and integration- We scaled the single stand management model (Chapter 4) by allowing it to simulate the effects of management over the estimated mature forest area available in the watershed. At each year of the simulation, based on the management regime specified, the model calculates how much forest is needed to

supply the timber industry demand. While the model is not spatially explicit, it efficiently simulates the process of large contiguous forests subdivided into stands with diverse patterns of use (Figure 6-2). Consequently, the watershed model does not consider distance-based costs. However, marginal river transportation costs are low and are not a major cost factor in firm operation (Chapter 5). The firm economic model was simply scaled by multiplying single-firm inputs and outputs by the number of firms present in the watershed.

The starting condition of each simulation is a large forest area of structure and composition reflecting the ~40 ha of permanent inventory plots spread in recently undisturbed areas across the watershed (i.e., in areas with no sign of logging in the last 20 years). This is likely an optimistic assumption as timber has been continuously harvested in the watershed. To partially counter this bias, from the original mature forest area we subtracted a degradation proportion equivalent to assuming all mature within 100 meters of mapped rivers and streams are not manageable (14% of watershed) following the field observation that most forests along rivers commonly have few timber trees of commercial value (e.g., older timber-poor secondary forests, heavily logged forests). The importance of this degradation parameter was later assessed in our model sensitivity analyses.

Once the total available management area was determined, this area was divided into two hypothetical stands, one owned by firm owners, calculated as the number of firms times average property size derived from Pinedo-Vasquez et al. (2001), and stands owned by the other approximately 100 smallholder families residing in the watershed. At each yearly time step, previously harvested stands old enough for more

harvests (determined by the management cutting cycle), can have their area entirely or partially harvested again if necessary. To reflect local behavior, firms will preferentially harvest firm-owned stands to avoid stumpage costs.

At each level of production, the model calculates the mean volume per tree harvested based on total volume and number of individuals harvested in all harvested areas. Mean volume per tree influences extraction costs and stumpage costs as firms purchase standing timber per tree. Lastly, stumpage costs are calculated using mean volume per tree values only for the proportion of trees harvested outside of firm-owned areas. All resulting watershed simulations used a 50-year time horizon to balance between short run economic assumptions and long-term ecological consequences of management.

Model Optimization and Stochasticity

Given açai fruit production is a common alternative to timber production in the Amazon Estuary (Anderson 1986, Munizmiret et al. 1996), we created a simulated annealing optimization that finds the optimal yearly timber harvests for firms in the industry to maintain a comparative worth-while investment. At the start of each simulation the model calculates the NPV/startup costs ratio of açai fruit production in the watershed for a 50-year period (Chapter 5). This ratio is then used to estimate the necessary NPV equivalent for timber production based on average firm startup costs. To ensure NPV of timber firms equal the açai equivalent at the end of the simulation, the model optimizes industry production each year such that

$$\text{Revenue} - \text{Costs} - \text{Annualized startup costs} = \text{Annualized Açai NPV equivalent} \quad 6-1$$

The açai NPV equivalent and startup costs were annualized using an equivalent future annual payment formula further discounted for 1 year for present value (Klemperer 1996; equation 6-2) where V_0 is present value, r is discount rate, and n the number of payment years. These annualized values were necessary to find the industry yearly production that yielded the desired NPV at the end of the 50-year simulation.

$$p = \frac{r \times [V_0 \times (1+r)]}{[1 - (1+r)^{-n}]} \quad 6-2$$

The yearly optimization considers industry production volume between 0 and maximum potential production based on sawmill processing capacity but is also bounded by watershed available timber. The model selects the industry production level that will most closely satisfy the above objective function, including no production when marginal costs are greater than marginal revenue. While we do not assume real smallholder firms perform a similarly complex calculation, this optimization is useful to evaluate whether timber production is a competitive land use option for the Mazagão watershed.

Once the base industry simulation model was completed, we incorporated uncertainty of underlying costs, revenues and production parameters to perform a viability analysis of the entire industry over the projected simulation period. Random values for proportion variables (e.g., sawmill efficiency) were modeled using the beta distribution. All other non-zero parameters had random values drawn from a normal distribution truncated at zero (e.g., mean days of sawmill operation per month). At the beginning of each stochastic run, the model draws values for each economic parameter based on their mean values and standard errors. For any given watershed management scenario (*described below*), industry viability was calculated as the proportion of its

10,000 stochastic runs that had average firm NPV above the açai NPV equivalent for optimization scenarios or zero for business-as-usual scenarios.

Watershed Scenarios

A total of 10 watershed scenarios considered variation in management regimes, local industry timber consumption and informal and legalized modes of production (Table 6-1). Forest management regimes included extraction with no management, federal management as required by Brazilian law (FDM), and the optimal sustained timber management regime (STM) based on long-term projections of population growth and timber yield for the region (Table 6-1; Chapter 4). The no-management scenario was used to simulate current common practices in the watershed and is based on extensive field monitoring and interviews (Chapter 4 and 5). It assumes small minimum harvest size, no commercially viable seed trees (i.e., 100% harvest intensity), and a very short harvest cycle that approximates continuous re-entry logging. While not all watershed inhabitants, especially some long-term residents, allow such levels of intense harvests in their private plots, most inhabitants harvest or sell standing timber without concern for tree size, species used or remaining stock. Our federal management scenarios may lead to conservative harvests since the Brazilian Federal Law includes a special clause that allows harvests of 3 trees per ha in floodplain forest areas (instead of the standard 10 m³ per ha limit). However, we did not consider this special clause in our model since its approval is done in a case-by-case basis, and depends on volumetric studies conducted by the agency enforcing the law. Furthermore, we were interested in the effectiveness of the federal law that applies to more than 95% of timber harvests in the Amazon (Lentini et al. 2005). Watershed scenarios varied in yearly timber industry consumption by either including constant business-as-usual (BAU)

timber consumption rates derived from local firm monitoring or dynamic consumption rates based on our NPV-optimizing routine.

Since estuarine smallholder firms operate informally without regards to Brazilian environmental, labor and fiscal laws (Chapter 5, Barros and Uhl 1995), we created watershed scenarios that included the additional costs of legalization of timber production. These costs include inventory costs for each year's harvest area, costs of management plan elaboration, wage legal burden (payroll taxes and related legal obligations), costs of mandated safety equipment, training costs and state sales tax. The wage legal burden, inventory, safety and training costs were based on inflation-discounted values from Holmes et al. (2002). Management plan elaboration costs were based on consultations with the Brazilian Forest Service and companies that provide the service to small- and community-sized operations (\$16.84/ha or \$31.09/ha if including inventorying). Lastly, since smallholders operating under informality generally sell timber at low prices (Chapter 5, Scherr et al. 2004), we included a timber sale price adjustment for all scenarios pondering legalization of timber production. This legal timber price was estimated from the sale value of floodplain forest timber by a community-forestry project in Amazonas (Humphries 2010). The influence of legal timber price on industry viability was explored in subsequent sensitivity analyses.

Model Outputs

For each watershed scenario the model created multiple outputs including species size distributions for each resulting stand, detailed per year costs and revenues. To quantify and contrast the ecological consequences of the local timber industry on watershed's forests, for each simulation the model calculated population growth rates for species (λ), the proportion of available forest area harvested, total volume

harvested, the average number of harvests per ha of available forest area and total harvested area. To differentiate changes in forest structure due to contrasting management, the model also calculated the proportion of initial available area experiencing a 50% decline in stand structure parameters including total number of individuals, harvest species volume, number of large trees, number of large commercial grade trees.

Besides simple elasticity and correlation analyses between model parameter and model outcome to evaluate model performance, we chose a select number of variables of interest for a detailed sensitivity analysis: interest rate; species used; area owned by firms; number of firms in watershed; proportion of area degraded; sawmill efficiency; and timber price. The importance of these variables was explored by changing them to a set of values representing a their likely range and including values of particular interest (Table 6-2).

Results

Scenario Outcomes

What are the prospects for the Mazagão timber industry operating informally under current levels of timber demand? (Scenarios 1, 3, 7)

Model simulations show that NPV is high for the industry when operating informally under current levels of timber demand, whether without management, FDM or STM (S1, S3, S7; Table 6-3). Average firm NPV was highest for STM (\$47,555) and FDM (\$36,669) compared to no management (\$23,966). Due to per-area harvest volume limits, the FDM scenario leads to the entire available area in the watershed being harvested every cutting cycle of 10 years (i.e., 5 harvests per ha of manageable forest in 50 years). The STM scenario also leads to the entire watershed being

harvested, albeit under a less frequent regime (3.5 harvests per ha). In contrast, under no management only 47% of the available watershed forest is harvested. However, these intense harvests lead to worse ecological consequences in terms of reduced average population growth rates and all areas harvested showing substantially altered population structure at the end of the 50-year projection period. While both STM and FDM showed stable or increasing population growth and no area being significantly structurally altered, the STM scenario outperformed the legal scenario in terms of profitability and greater protection to populations with few individuals. Consequently, stochastic simulations show that the probability of long-term industry viability is largest for STM (99%) but closely followed by FDM (95%). The lack of management drops the probability of industry persistence to 0.77%.

What are the prospects for industry operating informally seeking financial returns similar to Açaí fruit production? (Scenarios 2, 5, and 9)

Given average firm start up costs, firms would need a NPV of \$26,300 to have a similar profitability to açaí fruit production. Model results show this is only possible if the industry uses FDM or STM guidelines. Without management, when the industry seeks higher yearly profits it runs out of unharvested areas within 15 years and must reharvest areas not yet recovered from recent intense harvests. Under no management, harvesting trees down to small sizes also lead to more costly future harvests as stumpage paid per tree yields less volume to be processed and sold. The final result is that without management the industry crashes yielding a low average NPV of \$4,402, with several years of no production due to high marginal costs and the entire management area being significantly affected structurally (Table 6-3). In contrast, by

using either FDM or STM guidelines, timber production remains profitable compared to extraction with no management during the 50-year projection while keeping watershed forests structurally healthy. However, the STM scenario leads to slightly better results since it requires less total volume (3.1×10^5 vs 3.5×10^5 , respectively) and reduced number of harvests per ha (2.07 vs 4.20, respectively). Consequently, the probability firms in the industry will keep an average NPV equal or higher to \$26,300 is highest under STM (0.86), followed by FDM (0.52), and very low if the industry uses no management (0.05).

What is effect of legality on the viability of timber production by Mazagão smallholders? (Scenarios 4, 6, 8,10)

All legal scenarios exhibited similar legalization costs. Using the FDM BAU scenario as example, average yearly costs of legalization per firm was 11% (\$1,905) of average yearly operational costs. This cost per firm was divided among management plan elaboration (\$1,184.88), inventorying (\$296.23), training (\$245.39), block layout (\$153.24), and safety (\$25.49). Legal costs per ha of harvests was \$27.08 and would be slightly higher for the Mazagão industry if it opted to hire consultancy firms that also conduct all necessary fieldwork (\$31.45/ha).

The added legalization costs changed the profitability of all management scenarios. Under BAU production and FDM, average firm NPV goes from \$36,669 to -\$2,429, making timber production an unprofitable investment. STM under BAU also suffers a large decrease in NPV by legalization, but fares better than FDM by still resulting in a positive NPV (\$15,894). Consequently, if firms continue to produce at similar levels to current volumes, the probability of industry persistence under legality is highest for STM (0.73) than FDM (0.27). When optimizing for açai NPV equivalence

under legalization, FDM only reaches a NPV slightly above 0 falling far short of the necessary \$26,300 value and leading to many years without production. Legal STM also fails to reach NPVs equivalent to açai, but reaches an NPV much higher than FDM while avoiding years without production (\$21,489).

Consequences of Changing Factors of Production

Interest rate- As expected, increasing interests rates from the current 6% to 25% leads to drastic reductions in NPV for all scenarios. Apart from the optimized legal FDM scenario, NPVs of all scenarios remained positive even at the highest interest rate.

Species- The exclusion of *M. paraensis* from harvests is catastrophic for all watershed scenarios. For scenarios with no management, had the industry not changed recently to use *M. paraensis*, it would face a bleak future with all available forests harvested, NPV near zero, and a large number of harvests per ha leading to all watershed forests suffering large structural changes by the end of the simulations. For FDM, NPV drops substantially in all cases but only down to negative values when combined with legalization (S4 and S6). Exclusion of *M. paraensis* from FDM scenarios also leads in most cases to an increase of years without production, but no long-term structural consequences. Due to higher DCM, SFM scenarios are entirely dependent on *M. paraensis* dominant share of large trees. Without the species, NPV drops from generally high values to below zero with most years without production due to timber shortages. In contrast, the exclusion of the fast growing but currently scarce *V. surinamensis* has little influence on most scenario outcomes.

Area owned by firms- In BAU scenarios, increases in firm-owned forest area increases average NPV, allowing NPV to be positive for FDM legal BAU scenario (S4) when firm area exceeds twice the current estimate (54 ha). Except for that scenario,

firms can operate with a positive NPV even with no firm-owned area, suggesting that firms are not entirely dependent on timber from own properties to maintain profitability. In optimization scenarios where firms failed to be as profitable as açai palm fruit production (S2, S6, S10), increasing firm-owned forest area to 200 ha (a size larger than all but one property in the watershed) was still not enough to make timber production as profitable as açai fruit production. For optimization scenarios where açai NPV equivalence was reached (S5 and S9), additional firm-owned area allowed for reducing total volume harvested and the number of harvests per ha for watershed forests.

Number of firms- Surprisingly, average NPV remains unchanged with an increased number of firms in the watershed under the informal BAU scenario with no management. However, number of harvests per ha and proportion of area suffering large structural changes increase with each additional firm (Figure 6-3). For all other FDM and STM BAU scenarios, NPV falls gradually with an increasing number of firms in the watershed. For scenarios with management, despite the decreases in NPV and increase in number of harvests per ha with an increase in number of firms, only a small portion of watershed forests (<10%) suffers major structural changes by the end of simulations.

Percent of area degraded- For nearly all BAU scenarios, with increasing degraded forest area not immediately available for management, NPV decreases as the number of years without operation increase due to timber shortages. Only the no management informal scenario has average NPV immune to decreasing forest area available to management (S1). However, in this scenario, with increased area degraded

remaining areas are more frequently harvested and proportion of watershed suffering large structural changes increase. Similarly to BAU scenarios, under optimizing scenarios NPV decreases and number of shortage years increases with area degraded, but with STM scenarios being less affected than FDM and no management scenarios. As with other variables, forests suffer little to no large structural changes under FDM or STM at all levels of watershed degradation.

Mill efficiency- Because mill processing was the large contributor to total operational costs, mill efficiency had one of the largest impacts on scenario viability. For all BAU scenarios, average firm NPV is positive at the high efficiency value representative of average sawmill efficiency in the Amazon (Lentini et al. 2005), including those incorporating legalization costs. For nearly all optimization scenarios, at the low mill efficiency level representative of average efficiency for all micro firms of the Estuary (Lentini et al. 2005), marginal costs exceed marginal revenues resulting in nearly no harvests for most years. Above the average mill efficiency observed in the watershed, optimization scenarios resulted in fewer years without production and fewer harvests per ha.

Timber price- Especially relevant to the legalization scenarios, timber prices had a large effect on industry profitability. At the high price estimate (\$315.34 per m³ of sawn wood), NPVs of all scenarios were multiplied well above the açaí NPV equivalent of \$26,300. The small difference between current and legal price used (\$92.35 vs \$112.00 per m³ of sawn wood) was enough to make NPVs negative in all legal scenarios (S4, S6, S8, S10). For STM and FDM optimizing scenarios without legalization costs, average NPV remained above the açaí NPV equivalent regardless of

timber price. For optimization scenarios, with increasing price the necessary number of harvests per ha decreased steeply, changing, for instance, from 5 to <1 between low and high legal price for STM and FDM legal scenarios (\$112.00 vs \$315.34; S6 and S10).

Discussion

The Benefits of Forest Management

Our scenario simulations show that on a watershed/ local industry scale, forest management in general yields better ecological and economic outcomes than extraction with no management. Federal management guidelines provide a clear improvement over no management scenarios, but the STM guidelines designed for the Mazagão watershed in previous analyses (Chapter 5) fares even better. In general, management improves the prospects for long-term profitability, consequently increasing the probability of industry persistence in all cases. Conservation wise, management results in extensive harvests that allow for faster population recovery while preventing large changes in forest structure. As shown in the no management profit-optimizing scenario, a more aggressive profit seeking industry can lead to population crashes. In contrast, STM guidelines can be what makes legalization possible under current levels of production.

Importance of Model Parameters

Economic factors beyond forest management (e.g., price, sawmill efficiency) were very important in defining industry viability, bridging the difference in several management scenarios between industry failure and persistence. While these results illustrate the importance of the proper consideration of economic factors in efforts to expand the area under management in tropic forests, increased profitability alone is not

a recipe for SFM (Rice et al. 2001). Ecological factors (e.g., species used, watershed degradation proportion) also had a large effect on scenario outcomes. Partially due to the reduced diversity of flooded forests (Ferreira and Stohlgren 1999), the exclusion of the predominant species, *M. paraensis*, from watershed scenarios greatly reduced the probability of industry long-term viability. The importance of *M. paraensis* to the future prospects of the local industry is also likely tied to the long history of sequential timber species depletion in the region (Chapter 3). Given the species studied account for 62% of basal area of trees ≥ 30 cm DBH in Mazagão forests and that most other species seem to lack merchantable wood properties (Fortini, *unpublished data*), *M. paraensis* is one of Mazagão's last chance to make forest management work.

Industry Economic Viability

Our study supports past findings showing that great profitability is possible for individual firms operating informally in the watershed (Chapter 5). However, using a single-firm economic model detached from estimates of available forest area and species ecology can lead to overestimates of firm profitability and industry viability. In our study we tie economic production to limited timber resources, the forest's ability to regenerate, changes in harvests from shifts in species abundance and size distributions, and a dynamic estimate of timber produced from firm-owned forests. Under these restrictions, most scenarios where the industry attempts to achieve the same profitability as açai fruit production ran into the ecological limits of the watershed, a fact that would otherwise be concealed from a simple economic analysis. However, firm profitability aside, the labor-dependent nature of timber use by micro firms represent a major source of employment to a local work force that has few other economic opportunities (Chapter 5).

Industry Ecological Viability

The majority of past research in tropical forests addresses the issue of sustainability of use and management from a single stand perspective (Osho 1995, Gourlet-Fleury et al. 2005, Hao et al. 2005). Our results show that we must keep in mind the difference between stand and watershed/landscape ecological sustainability. A stand may have the potential to provide sustainable yields, but the watershed/landscape may still not have enough area to make a shift to STM possible. Ecologically, despite the sequential depletion of some timber species in the region (Chapter 3), stand conditions and favorable population dynamics still allow for a level of sustainable production (Chapter 4). Using our watershed-scale approach, with further degradation of watershed forests the ecological resource base necessary to ensure SFM is threatened (Figure 6-4). In that respect, our results show management is crucial to safe guard against further forest degradation. Finally, due to lack of long-term ecological monitoring data and computational limitations, our model only incorporates economic stochasticity, which likely underestimates the importance of underlying species ecology to long-term industry persistence.

The Consequences of Legalization

Legalization cannot be viewed simply as a costly bureaucratic environmental licensing requirement. Legalization offers benefits such as increased workforce safety and benefits and is currently tied to management guidelines that were shown to be very effective in improving ecological and economic viability of timber production in the watershed (Chapter 4). Nevertheless, the inadequacy of forest management legislation to smallholder and community enterprises is a long-standing root of informality in the sector (Kaimowitz 2003, Scherr et al. 2004). The additional costs of legalization

primarily from inventorying and management plan elaboration greatly reduced profitability and consequently lowered industry long-term viability of all scenarios with and without management. As a result, in scenarios where profits were optimized to reach equivalence to açai fruit harvests, forest use intensified greatly to compensate for larger costs (Figure 6-5). As an example, under the STM profit-optimizing scenarios harvests have to nearly double to compensate for extra legalization costs, lowering probability of industry persistence from 0.86 to 0.19 while yielding profits below those expected from açai fruit production. The issue of undervalued tropical timber is a known barrier to forest management adoption (Richards 2000). In our study, the viability of all legalization scenarios relied heavily on the legalized local industry having access to better prices than currently low timber prices.

Perhaps the largest challenge to legalization of timber use in Mazagão and similar watersheds is that, due to lack of viable mechanisms for smallholder legalization (d'Oliveira 2000, Scherr et al. 2004), legal scenarios assume a cooperative approach in which firms share legalization costs and coordinate their activities across multiple smallholder property boundaries. While progress is slowly being made in terms of legal support for community management in tropical forests, with 400,000 smallholders in Amazon forests alone, the lack of viable legal mechanisms for smallholder timber use is a glaring omission (Nepstad et al. 2009). In the few places where simplified management plans are required, care must be taken not to compromise long-term sustainability with potentially over-simplified management guidelines devoid of ecological principles (Keller et al. 2007).

Prospects for Management

Whether legalized or not, forest management in Mazagão should be of interest to forest users and conservationists alike, a finding that is clearly contrasted with a general lack of management in the region. The low rates of management adoption in tropical forests have mystified forest managers and ecologists (Putz et al. 2000). As our model only contemplates firm activity in the watershed, it does not reflect the pattern of firm mobility documented in upland Amazonian forests (Stone 1998). With firm mobility, even micro firms can move from watershed to watershed following a pattern of forest depletion. In fact, a few Mazagão firm owners have moved to the region in the last 10 years from timber-depleted islands down river. Unfortunately, as firms continue to operate without restraint, effective management area decreases thus reducing the prospect of future management. Under this self-feeding loop, the present industry (extracting timber informality and without management) can avoid the negative consequences of increasing forest degradation by harvesting remaining forests more frequently. In contrast, this decrease in effective management area will lead to the reduced probability of industry persistence if the industry tries to adopt management such as the federal management guidelines that require longer harvest cycles and per hectare harvest limits (Figure 6-6).

Our research demonstrates that conceptually simple interdisciplinary models can help to find management options and explore their projected consequences at a watershed/landscape scale. With better long-term monitoring of forest use and responsive forest managers, similar models could be a key component in adaptive management for SFM in the tropics (McGinley and Finegan 2003). As we show in our results, such models can lead to optimized management regimes that yield better

economic and ecological outcomes (Chapter 4) that consider regional-specific community composition and ecology (Chapter 2 and 3).

Our past research shows that the ecological potential for sustainability was likely present in Mazagão from the start (Chapter 2 and 3). However, this potential is hidden by a long history of mismanagement (Chapter 3), undervalued timber prices (Chapter 5) and lack of adequate legislation and government support. In that respect, the search for SFM is not simply a search for a hidden technical solution. While the ecologically-based management is a critical component of SFM, action must be taken that counters the myriad of factors beyond forest canopies that preclude SFM in the first place. The unfortunate irony is that increasing government efforts to restrict informal timber use in the region suggest timber production will likely be phased out within a few years, closing a door to a potentially sustainable model of forest management in the Amazon.

Table 6-1. Watershed management scenarios

Scenario	Management type	Industry output	Legal	Harvest intensity	Min cut diameter (cm DBH)	Min density (n/ha)	High quality seed trees	Length harvest cycle (years)	Per ha harvest limit (m ³ /ha)
S1	No_mgm-Inf-BAU	BAU	n	1	30	0	n	5	-
S2	No_mgm-Inf-Opt	Optimized	n	1	30	0	n	5	-
S3	Fd_mgm-Inf-BAU	BAU	n	0.9	50	0.03	n	10	10
S4	Fd_mgm-Leg-BAU	BAU	y	0.9	50	0.03	n	10	10
S5	Fd_mgm-Inf-Opt	Optimized	n	0.9	50	0.03	n	10	10
S6	Fd_mgm-Leg-Opt	Optimized	y	0.9	50	0.03	n	10	10
S7	STM-Inf-BAU	BAU	n	0.5	60	1	y	10	-
S8	STM-Leg-BAU	BAU	y	0.5	60	1	y	10	-
S9	STM-Inf-Opt	Optimized	n	0.5	60	1	y	10	-
S10	STM-Leg-Opt	Optimized	y	0.5	60	1	y	10	-

Table 6-2. Parameters and values used in model sensitivity analyses

Parameter	Values	Description
Interest rate	0.0679*, 0.1, 0.15, 0.2, 0.25	
Species used	All*, - <i>Mora</i> , - <i>Virola</i>	Excluded either <i>Mora</i> or <i>Virola</i> to assess the importance of the two most important yield species in the watershed
Area owned by firms	0, 27*, 54, 100, 200	
Number of firms in watershed	6, 9, 12*, 15, 18	
Proportion of area degraded	0.14*, 0.2, 0.3, 0.4, 0.5	From current low estimate of degradation to a high degradation estimate
Sawmill efficiency	0.28, 0.37*, 0.42	0.28- average for Amazonian micro firms; 0.42- average for Amazonian mills
Timber price (U\$/m3)	92.35, 112, 315.34	\$92.35- current timber value; U\$112- price floodplain community forest operation sell legal timber; \$315.34- Brazilian Forest Service price estimate for low grade timber

* parameter value for deterministic model

Table 6-3. Watershed management scenario outcomes

	S1 No_mgm-Inf-BAU	S2 No_mgm-Inf-Opt	S3 Fd_mgm-Inf-BAU	S4 Fd_mgm-Leg-BAU	S5 Fd_mgm-Inf-Opt	S6 Fd_mgm-Leg-Opt	S7 STM-Inf-BAU	S8 STM-Leg-BAU	S9 STM-Inf-Opt	S10 STM-Leg-Opt
Mean λ	0.993	0.995	1.000	1.000	1.000	1.001	1.000	1.000	0.999	1.000
Prop of available area harvested	0.47	1	1	1	1	1	1	1	1	1
Total volume harvested (M3)	4.5E+5	3.6E+5	4.2E+5	4.2E+5	3.5E+5	4.2E+5	4.5E+5	4.5E+5	3.1E+5	5.6E+5
Average firm NPV (U\$)	23966	4402	36669	-2429	26306	1378	47555	15894	26306	21489
Years without production	0	18	0	0	0	25	0	0	0	0
Largest interval without production	0	4	0	0	0	5	0	0	0	0
Mean # of harvests per ha of available area	0.81	2.37	5	5	4.2	5	3.51	3.51	2.07	4.83
Mean # of harvests per ha of harvested area	1.74	2.37	5	5	4.2	5	3.51	3.51	2.07	4.83
Prop available area w/ 50% \downarrow in population size	0.47	1	0	0	0	0	0	0	0	0
Prop available area w/ 50% \downarrow in volume	0.47	1	0	0	0	0	0	0	0	0.08
Prop available area w/ 50% \downarrow in n large trees	0.47	1	0	0	0	0	0	0	0	0
Prop available area w/ 50% \downarrow in n large commercial trees	0.47	1	0	0	0	0	0	0	0	0
Probability of long-term industry viability	0.77	0.05	0.95	0.27	0.52	0.01	0.99	0.73	0.86	0.19

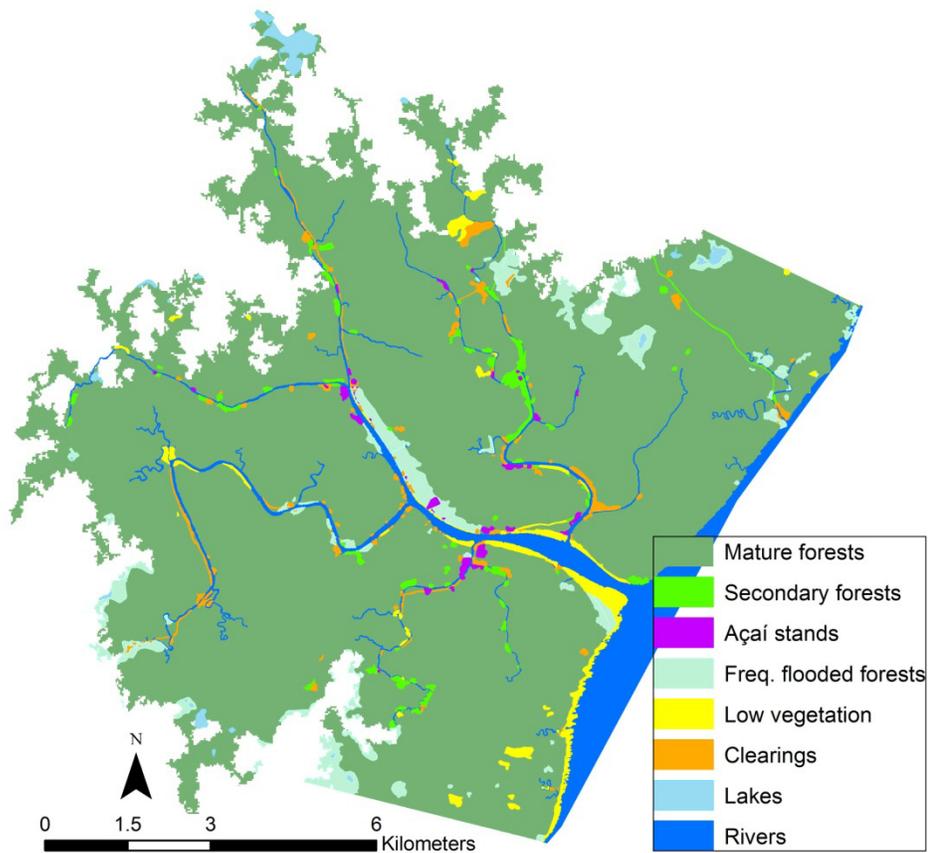


Figure 6-1. Map of Mazagão watershed landcover

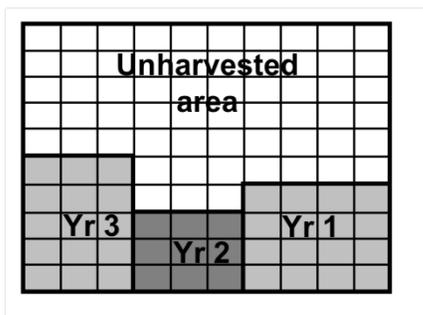


Figure 6-2. Graphical representation of simulated yearly harvests on watershed forests

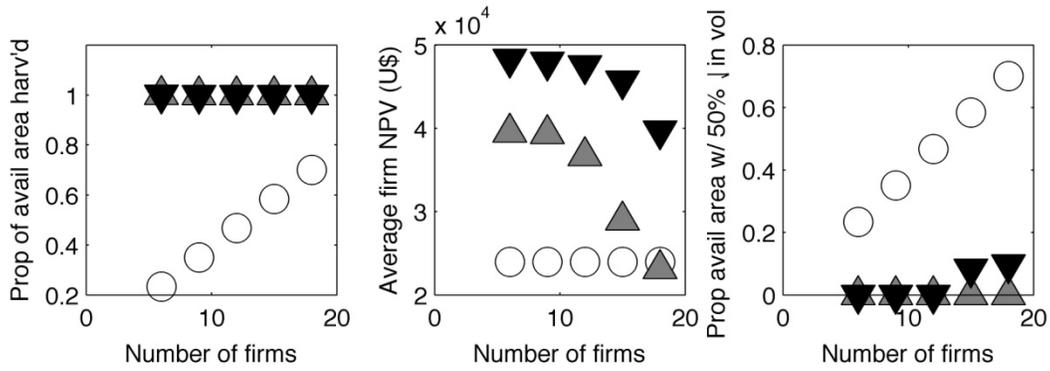


Figure 6-3. The consequence of local industry size to watershed scenarios with business-as-usual (BAU) informal production.

White- no management, grey- federal management guidelines (FDM), black- regionally derived sustainable timber management guidelines (STM).

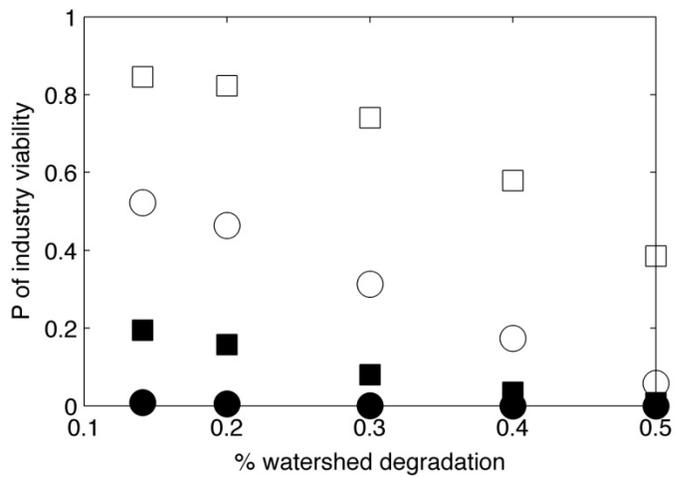


Figure 6-4. Decrease in the probability of industry persistence with increasing watershed areas unavailable for management under alternate profit-optimizing forest management scenarios.

Circles- FDM, squares- STM; White- informal production, black- legal production.

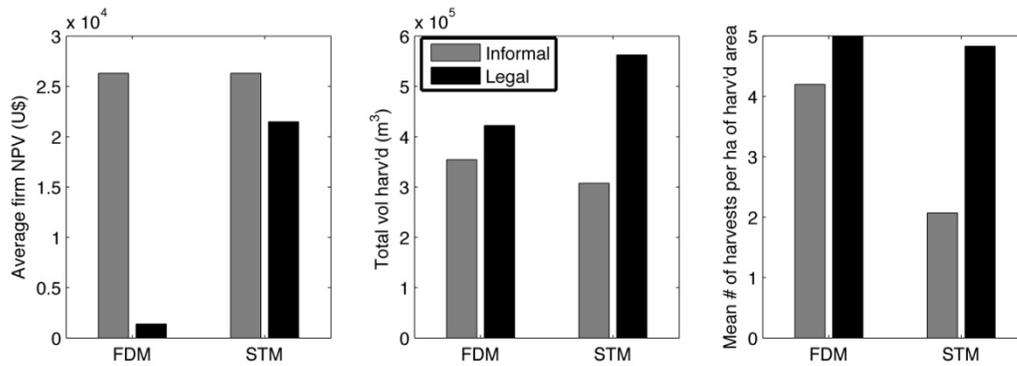


Figure 6-5. Consequence of adopting legalization for watershed scenarios with firms seeking açaí profit equivalence while adopting federal management guidelines (FDM) and regionally derived sustainable timber management guidelines (STM).

Note that only under informal production do profits reach the açaí equivalent value of US\$ 26,302.

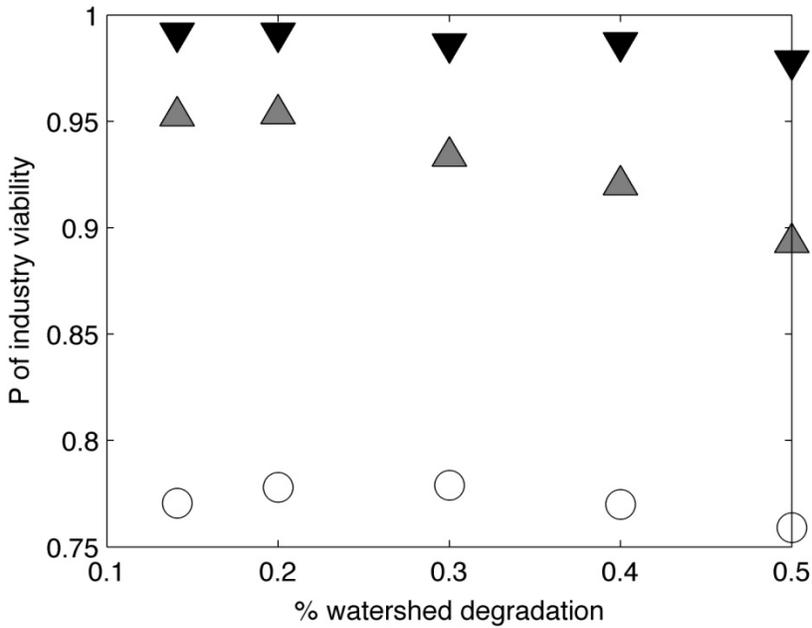


Figure 6-6. Decrease in the probability of industry persistence with increasing watershed areas unavailable for management under business-as-usual (BAU) informal production.

White- no management, grey- federal management guidelines (FDM), black- regionally derived sustainable timber management guidelines (STM).

APPENDIX
SPECIES LIST AND ABUNDANCE FOR MAZAGÃO AND IPIXUNA SAMPLE PLOTS

Abundance values reported as average number of individuals per hectare and standard errors. Abundance of understory individuals is reported in thousands of individuals per hectare.

Species	Common name	Family	DBH >= 5 cm		DBH >= 30 cm		Understory	
			Mazagão	Ipixuna	Mazagão	Ipixuna	Mazagão	Ipixuna
<i>Spondias mombin</i> L.	tapereba	Anacardiaceae	2 (1)	14 (4)	1 (0)	8 (2)	7 (0)	13 (1)
<i>Tapirira guianensis</i> Aubl.	tatapiririca	Anacardiaceae		12 (0)				
<i>Guatteria inundata</i>	envireira	Annonaceae	1 (0)					
<i>Guatteria poeppigiana</i> Mart.	envira preta	Annonaceae	4 (1)	10 (2)	1 (0)	2 (1)		13 (0)
<i>Ambelania acida</i>	pipeira	Apocynaceae	9 (0)					
<i>Aspidosperma desmanthum</i> Benth.ex.Mull Arg.	pau de arara	Apocynaceae	2 (1)	2 (1)	1 (0)			
<i>Malouetia tamaquarina</i> (Aubl.) A.DC.	molongo	Apocynaceae	1 (0)					
<i>Dieffenbachia cf. seguine</i> (L.) Schott.	aninga para	Araceae						13 (3)
<i>Philodendron</i> sp	tajá	Araceae						392 (48) 281 (137)
<i>Astrocaryum gynacanthum</i> Mart.	munbaca	Arecaceae						3 (0)
<i>Astrocaryum murumuru</i> Mart.	murumuru	Arecaceae	78 (7)	281 (28)		1 (0)	2968 (144)	9300 (488)
<i>Attalea excelsa</i> Mart.	urucuri	Arecaceae	7 (3)		5 (2)		60 (10)	
<i>Bactris maraja</i> Mart.	marajá	Arecaceae					70 (8)	251 (30)
<i>Euterpe oleracea</i> Mart.	açaí	Arecaceae	403 (84)	566 (164)			1538 (200)	1970 (100)
<i>Geonoma aspidiifolia</i>	ubim	Arecaceae					151 (15)	84 (0)
<i>Manicaria saccifera</i> Gaertn.	buçu	Arecaceae	3 (0)				50 (0)	
<i>Oenocarpus distichus</i> Mart.	bacaba	Arecaceae	3 (0)					
<i>Socratea exorrhiza</i> (Mart.) Wendl.	paxiuba	Arecaceae	2 (0)	7 (0)			20 (1)	64 (8)
<i>Tabebuia</i> sp	ceru	Bignoniaceae	1 (0)					
<i>Bombax munguba</i> Mart. et Zucc.	munguba	Bombacaceae	1 (0)	3 (2)	1 (0)	3 (2)	3 (0)	
<i>Ceiba pentandra</i> (L.) Gaertn.	samauma	Bombacaceae	1 (0)		1 (0)			
<i>Matisia paraensis</i> Huber.	cupuaçurana	Bombacaceae	8 (4)		1 (0)		7 (0)	
<i>Pachira aquatica</i> Aubl.	mamorana	Bombacaceae	4 (1)	1 (0)	2 (1)		7 (0)	
<i>Quararibea guianensis</i> Aubl.	inajarana	Bombacaceae	14 (5)				27 (3)	
<i>Protium</i> sp	breu	Burseraceae					3 (0)	382 (76)

<i>Protium pubescens</i> Ducke.	breu branco	Burseraceae	6 (2)	6 (1)		2 (0)	7 (0)	27 (3)
<i>Campsiandra laurifolia</i> Benth.	acapurana	Caesalpiniaceae	8 (3)	1 (0)	2 (1)			
<i>Crudia oblonga</i> Benth.	iperana	Caesalpiniaceae	2 (1)		2 (0)			
<i>Cynometra spruceana</i> Bth.	pau ferro	Caesalpiniaceae	2 (1)	1 (0)	2 (0)		7 (0)	
<i>Macrolobium pendulum</i> Willd. Ex. Vog.	ingarana	Caesalpiniaceae	11 (7)	2 (0)			13 (0)	40 (5)
<i>Mora paraensis</i> Ducke.	pracuuba	Caesalpiniaceae	23 (12)		7 (3)		472 (53)	
<i>Swartzia longsdorffii</i> Raddi	patroneira	Caesalpiniaceae		1 (0)				
<i>Swartzia racemosa</i>	pacapea	Caesalpiniaceae	28 (6)	1 (0)	5 (2)		479 (24)	
<i>Cecropia palmata</i> Willd.	embauba	Cecropiaceae	2 (1)	4 (2)		3 (1)		
<i>Licania heteromorpha</i> Benth.	macucu	Chrysobalanaceae	23 (8)	32 (5)	13 (5)	27 (4)	1082 (90)	429 (28)
<i>Licania kunthiana</i> H. F.	cariperana	Chrysobalanaceae	2 (0)					
<i>Licania macrophylla</i> Benth.	anoera	Chrysobalanaceae	15 (10)		4 (3)		50 (0)	
<i>Parinari excelsa</i> S.	parinari	Chrysobalanaceae		6 (2)		4 (2)	3477 (1050)	328 (157)
<i>Calophyllum brasiliense</i> Cambess.	jacareuba	Clusiaceae	2 (2)	3 (1)		3 (0)		20 (0)
<i>Caraipa grandiflora</i> Mart.	tamaquare	Clusiaceae	2 (1)		1 (0)			
<i>Clusia grandiflora</i>	cebola brava	Clusiaceae	7 (3)					
<i>Licaria mahuba</i> (Kuhl & A. Samp) Kosterm.	anani	Clusiaceae	2 (1)				201 (11)	10 (0)
<i>Rheedia macrophylla</i> (Mart.) Pl. et Jr.	bacuri pari	Clusiaceae		1 (0)				40 (3)
<i>Buchenavia ochroprumna</i> Eichler	tapacu	Combretaceae		3 (2)				
<i>Combretum cacoucia</i> Excell&Sandw	ioioca	Combretaceae	34 (8)				657 (62)	
<i>Terminalia dichotoma</i> G. Meyer	cuilarana	Combretaceae		1 (0)		1 (0)		
<i>Terminalia guianensis</i> Aubl.	cinzeiro	Combretaceae	4 (3)		4 (3)			
<i>Commelina</i> sp	maria mole	Commelinaceae					844 (31)	
<i>Tetracera</i> cf. <i>willdenowiana</i>	cipo de fogo	Dilleniaceae	7 (2)					
<i>Alchornea glandulosa</i> Poepp. E.Endl.	tamanqueira	Euphorbiaceae		1 (0)		1 (0)		
<i>Croton</i> sp	pau santo	Euphorbiaceae		1 (0)				
<i>Hevea brasiliensis</i> Mull. Arg.	seringueira	Euphorbiaceae	5 (1)		3 (1)		114 (6)	
<i>Hura crepitans</i> L.	assacu	Euphorbiaceae	4 (0)		2 (0)			
<i>Mabea pulcherrima</i> Muell. Arg.	faveira	Euphorbiaceae	2 (1)		2 (1)		7 (0)	
<i>Manihot subscandens</i>	maniva de veado	Euphorbiaceae					7 (0)	
<i>Sapium lanceolatum</i> Hub.	curupita	Euphorbiaceae		2 (1)		3 (1)		
<i>Ormosia macrocalyx</i> Ducke	tento	Fabaceae	1 (0)	3 (2)	1 (0)	1 (0)		20 (3)
<i>Ormosia smithii</i> Rudd	tento branco	Fabaceae		2 (0)				
<i>Platymiscium filipes</i> Benth.	macacauba da várzea	Fabaceae	8 (3)	3 (1)	5 (2)	2 (0)	57 (18)	3 (0)

<i>Pterocarpus amazonicus</i> Huber.	mututi	Fabaceae	11 (3)	2 (1)				1866 (179)
<i>Pterocarpus officinalis</i> Jacq.	mututirana	Fabaceae	3 (1)	3 (1)				7 (0)
<i>Ilex</i> sp	catingueira	Fagaceae	38 (14)	1 (0)				
<i>Banara guianensis</i> Aubl.	andorinha	Flacourtiaceae		1 (0)				
<i>Hernandia guianensis</i> Aubl.	ventoza	Hernandiaceae	1 (0)					
<i>Salacia</i> sp1	cipo capa bode	Hippocrateaceae	4 (1)					
<i>Salacia</i> sp2	cipo castanha de vô	Hippocrateaceae	1 (0)					
<i>Saccoglotis guianensis</i> Benth. var. <i>guianensis</i>	uxirana	Humiriaceae	2 (1)	2 (1)	1 (0)	1 (0)		
<i>Dendrobangia boliviana</i> Rusby	caferana	Icacinaceae	4 (2)		2 (1)			302 (141)
<i>Emmotum fagifolium</i> Desv. ex Hamilt.	muiraximbé	Icacinaceae		1 (0)				
<i>Poraqueiba sericea</i> Tul.	magonçalo	Icacinaceae	1 (0)		1 (0)			
<i>Aniba puchury-minor</i> (Mart) Mez.	louro amarelo	Lauraceae	2 (1)		1 (0)			
<i>Licaria mahuba</i> (Kuhlm & A. Samp) Kosterm.	mauba	Lauraceae	5 (2)	4 (1)	2 (1)	3 (1)	107 (11)	101 (21)
<i>Ocotea canaliculata</i> (Rich.)Mez.	louro pimenta	Lauraceae		1 (0)				
<i>Ocotea</i> sp	louro	Lauraceae	3 (1)	10 (9)		3 (0)	171 (13)	20 (7)
<i>Ocotea</i> sp	louro preto	Lauraceae	1 (0)	2 (1)		2 (1)		
<i>Ocotea</i> sp	louro branco	Lauraceae	2 (2)					
<i>Eschweilera tenuifolia</i> (Berg.) Miers.	matamata	Lecythidaceae	1 (0)					
<i>Gustavia augusta</i> L.	jeniparana	Lecythidaceae		24 (0)			60 (0)	127 (0)
<i>Vyrsonima</i> sp.	muruci	Malpigiaceae	1 (0)					
<i>Calathea ornata</i> Group.	cauaçu	marantaceae						13 (3)
<i>Ischnosiphon leucophaeus</i> (P&E)Koern.	guarumã canela	Marantaceae						1042 (187)
<i>Mouriri acutiflora</i> Naud.	camutim	Melastomataceae	7 (2)	4 (3)			17 (1)	27 (3)
<i>Carapa guianensis</i> Aubl.	andiroba	Meliaceae	22 (5)	1 (0)	11 (3)			181 (8)
<i>Carapa</i> sp	andiroba jaruba	Meliaceae	1 (0)					
<i>Cedrela odorata</i> L.	cedro	Meliaceae	1 (0)	1 (0)			3 (0)	10 (0)
<i>Guarea</i> sp	culhão de mucura	Meliaceae	2 (0)					
<i>Trichilia paraensis</i> C.DC.	jatauba	Meliaceae	3 (1)	1 (0)	1 (0)	1 (0)	10 (0)	
<i>Trichilia surinamensis</i> (Miq.) C.DC.	marajoão	Meliaceae	30 (8)	17 (8)	10 (4)	1 (0)	482 (40)	1149 (195)
<i>Inga alba</i> (SW) Willd.	inga xixica	Mimosaceae	1 (0)	6 (4)			3 (0)	452 (0)
<i>Inga cayenensis</i> Benth.	inga peludo	Mimosaceae					3 (0)	
<i>Inga cinnamomea</i> Spruce ex. Benth.	inga branco	Mimosaceae		20 (0)		6 (0)		10 (2)
<i>Inga edulis</i> Mart.	inga cipo	Mimosaceae					7 (0)	
<i>Inga lenticifolia</i> Benth.	inga preto	Mimosaceae		17 (6)				87 (0)

<i>Inga pilosula</i> (L.C.Rich) Macb	inga cavalo	Mimosaceae		3 (0)					
<i>Inga</i> sp	inga	Mimosaceae	3 (1)	7 (4)		2 (0)	50 (7)	499 (35)	
<i>Pentaclethra macroloba</i> (Willd) O. Kutzen	pracaxi	Mimosaceae	61 (9)		13 (4)		372 (27)		
<i>Pithecellobium inaequale</i> (H.B.K.) Benth.	jaranduba do mato	Mimosaceae	1 (0)				7 (0)		
<i>Stryphnodendron guianense</i> (Aubl.) Benth.	angelim da varzea	Mimosaceae		1 (0)		1 (0)		3 (0)	
<i>Zygia juruana</i> (Harms) L. Rico	inga cururu	Mimosaceae	29 (6)				60 (6)		
<i>Siparuna guianensis</i> Aubl.	capitiu	Monimiaceae		2 (0)					
<i>Chlorophora tinctoria</i> (L.) Gaud.	cutiti	Moraceae	2 (1)						
<i>Ficus anthelminthica</i>	quaxinguba	Moraceae	2 (0)	3 (1)		1 (0)			
<i>Ficus pertusa</i> C.F.	apui	Moraceae	3 (1)	10 (3)	2 (1)	3 (1)			
<i>Olmedia caloneura</i> Huber.	muiratinga	Moraceae	2 (1)	3 (1)	1 (0)	3 (1)	13 (0)	291 (62)	
<i>Sorocea duckei</i> W. Burger	camurim	Moraceae	2 (1)		1 (0)				
<i>Virola surinamensis</i> (Rol.) Warb.	virola	Myristicaceae	7 (2)	10 (5)	2 (1)	3 (1)	268 (26)	201 (24)	
Unknown sp	jeju	Myrsinaceae	1 (0)	4 (2)			3 (0)	228 (22)	
<i>Calyptranthes speciosa</i> Sagot.	goiabarana	Myrtaceae	7 (6)	53 (19)	1 (0)			1203 (72)	
<i>Pariana campestris</i> Aubl.	taboquinha	Poaceae					2502 (294)		
<i>Triplaris surinamensis</i>	taxi	Poligonaceae	7 (2)	2 (0)	1 (0)	2 (0)	13 (0)		
<i>Polybotrya caudata</i> Ktze.	samambaia	polipodiaceae					258 (27)	3 (0)	
<i>Alibertia</i> sp	ginja	Rubiaceae	2 (1)						
<i>Bothriospora corymbosa</i> (Benth) Hook.	pau macaco	Rubiaceae	1 (0)						
<i>Callycophyllum spruceanum</i> Benth.	pau mulato	Rubiaceae	58 (7)	33 (24)	45 (5)	12 (4)			
<i>Chimaphys barbata</i> Ducke	canela de velho	Rubiaceae						312 (0)	
<i>Genipa americana</i> L.	genipapo	Rubiaceae		1 (0)		1 (0)			
<i>Palicourea cf. calophylla</i> DC.	café bravo	Rubiaceae					3 (0)		
<i>Zanthoxylon</i> sp	laranjinha	Rutaceae	40 (12)		2 (0)		539 (24)		
<i>Allophylus amazonicus</i> (Martius) Radlkofer	olho de tucano	Sapindaceae	1 (0)						
<i>Talisia</i> sp	gogo guariba	Sapindaceae	1 (0)	3 (0)					
<i>Crysophyllum argenteum</i> subsp <i>auratum</i> (miq.) Penn.	guajara branco	Sapotaceae		9 (2)		3 (1)		121 (7)	
<i>Crysophyllum excelsum</i> Huber.	guajara	Sapotaceae		24 (14)		2 (0)		1625 (128)	
<i>Crysophyllum</i> sp1	guajara grande	Sapotaceae		2 (1)		2 (1)		47 (0)	
<i>Crysophyllum</i> sp2	guajara vermelho	Sapotaceae		1 (0)				3 (0)	
<i>Pouteria caimito</i> (prev. <i>laurifolia</i> Mart.)	abiu seco	Sapotaceae						3 (0)	
<i>Pouteria sagotiana</i> (Baill) Eyma.	maçaranduba da varzea	Sapotaceae	7 (1)		2 (1)		27 (10)		
<i>Pouteria</i> sp	abiurana	Sapotaceae	5 (3)		4 (3)		208 (37)		

<i>Sarcaulus brasiliensis</i> (ADC) Gyma	jarai	Sapotaceae	11 (3)	2 (1)	104 (8)	
<i>Simaba multiflora</i>	caximbeira	Simaroubaceae	14 (9)	1 (0)		
<i>Guazuma ulmifolia</i> Lam.	mutamba	Sterculiaceae	1 (0)			
<i>Herrania mariae</i> (Mart.) Dene.	cacau jacare	Sterculiaceae	2 (1)	3 (1)	17 (5)	74 (2)
<i>Sterculia speciosa</i> K.Schum	caputeiro	Sterculiaceae	2 (1)	4 (2)	1 (0)	1 (0)
<i>Theobroma cacao</i> L.	cacau	Sterculiaceae	3 (1)	28 (10)	10 (0)	34 (2)
<i>Apeiba aspera</i> Aubl.	penete de macaco	Tiliaceae	3 (1)	1 (0)		
Unknown sp	areu areu			6 (2)	1 (0)	
Unknown sp	caripé		1 (0)			
Unknown sp	inga amarelinho			1 (0)		
Unknown sp	marupá		1 (0)			
Unknown sp	bariri				881 (70)	
Unknown sp	isqueiro			2 (0)	1 (0)	
Unknown sp			2 (1)	2 (0)	1 (0)	13 (0)
Unknown sp				1 (0)		

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BIOGRAPHICAL SKETCH

Lucas Fortini was born in Rio de Janeiro in 1975. In 1993 he moved to California where he eventually got his A.A. degree in environmental studies from Moorpark College and his B.S. degree in natural resource management from UC Berkeley. Following his life-long interest in the natural environment, Lucas has been interested in how the response of populations and communities to natural and human-made disturbance determine ecological resilience and shape opportunities for conservation and management. After graduating in Berkeley, Lucas decided to focus his research in the tropics. Hired as a research scientist by Daniel Zarin at the UF, he spent a few years working on multiple forest ecology and management research projects in the Brazilian Amazon before starting his PhD. After his PhD, Lucas is driven to make crucial tropical forest conservation and management efforts more science-based by pursuing analytical and modeling approaches that yield relevant and applied results. A Brazilian, American, and Italian citizen, he holds his wife, nature and music as his life's passions.