COMMUNITY-BASED TIMBER MANAGEMENT IN ACRE, BRAZIL, AND ITS IMPLICATIONS FOR SUSTAINABLE FOREST MANAGEMENT

By

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This thesis is dedicated to my friends and colleagues in PAE Porto Dias, who welcomed me into their homes, and to my husband, Chris Baraloto, for coming this far in the journey together.

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In recent years, there has been much debate about whether conservation goals can be achieved by logging old-growth tropical forests. Even so, the global demand for tropical hardwood is not likely to subside anytime soon, making good management of forests outside of parks and other protected areas essential. Some researchers have suggested that rural communities may be the most ideal candidates to engage in sustainable timber management, since they generally harvest at low intensities. Yet, it has also been noted that the impact of natural resource exploitation at this scale is still not well known, and that sustainable forest management (SFM) guidelines will necessarily vary, depending on the location in which they are being implemented.

Field studies were conducted in Acre, Brazil to assess the impacts of the Porto Dias community timber management project. The main objectives of this investigation were (1) to determine the extent of area disturbed (treefall gap and skid trail construction) in a small-scale certified logging operation, (2) to determine a difference in damage incidence

to future crop trees (FCTs) ≥ 20cm dbh between forest with or without bamboo (*Guadua spp.*), (3) to determine a difference in damage incidence to FCTs ≥ 20cm dbh between those associated with lianas and those that are not, (4) to determine to what extent harvesting can be conducted more intensely (m^3ha^{-1}), without incurring greater damage to FCTs ≥ 20cm dbh, and (5) to determine to what extent marking of FCTs diminishes damage during logging. Related to this experimental study, an examination of key reduced-impact logging (RIL) harvesting techniques and their potential influences on population dynamics of tropical tree species in small scale timber operations was also initiated, with the goal of addressing potential future management strategies for the smallholder system.

The results from the study indicate that although the effects of marking, liana presence and location in *Guadua*-dominated forest were not significantly associated with FCT damage, the high values reported for area damaged merit attention. Previous studies have demonstrated that low intensity harvests are often associated with increased damage levels per tree due to skid trails, suggesting that the gap mosaic of the forest will often change, even when low-intensity RIL guidelines are followed. Given the series of poorly understood parameters outlined in this thesis, it is recommended that systematic post-harvest monitoring accompany logging operations, so that the benefits of recommended practices may be evaluated by managers at the appropriate scale.

CHAPTER 1
GENERAL INTRODUCTION

Deforestation is one of the greatest environmental challenges of the past few decades (Food and Agricultural Organization (FAO) 2005). Brazil, which claims 63% of the Amazon basin (Lentini et al. 2003), has long been at the center of the tropical conservation debate. The Brazilian Amazon constitutes approximately one-third of the world's extant tropical forests (Uhl et al. 1997). Since the 1960s, however, the country has been plagued by destructive development policies that have opened up the region to industrial logging, mining, ranching, and large-scale colonization projects, contributing to a 14% forest cover loss to date (Nepstad et al. 2002). Compounding the problem of forest cover loss, Brazil is simultaneously the world's largest producer and consumer of tropical timber, most of which is generated through land clearing or unplanned selective logging (Verissimo and Barreto 2004).

Selective logging is the harvesting method in which individual trees (rather than an entire group) are removed. When conducted on a rotational basis, selective logging is generally regarded to have a low impact on the forest (Uhl and Buschbacher 1988, but see Sist et al. 2003 a,b, Asner et al. 2005). Unplanned selective logging, however, typically excludes the use of a systematic timber inventory or management plan and often harvests the same area over short, repeated intervals, frequently removing high volumes of timber. The primary goal in this case is to extract the largest and most valuable commercial species with little regard to surface area damaged or future harvests (Schulze 2003). In many instances, this has led to acute (up to 50% or more) canopy reduction,

severely disturbed mineral soils and damage to the residual stand (Uhl and Vieira 1989, Verissimo et al. 1992, Johns et al. 1996). In response, scientists and practitioners alike have proposed implementing sustainable forest management (SFM) practices that could limit canopy cover loss and protect ecosystem function and biodiversity while maintaining the economic value of the forest (Putz 1994, 1996, Pearce et al. 2003). Definitions of SFM differ, but the term generally implies the sustenance of many forest products and services over long periods of time (Pearce et al. 2003). Yet, despite the recent conservation interest in SFM, there are many who would argue that to link conservation with "sustainable forest management" is erroneous, and that the only way to curb deforestation is outright protection (Rice et al. 1997, Bowles et al. 1998, Terborgh 1999).

Recently, there has been a massive undertaking in Brazil to reduce the negative environmental impacts of regional silvicultural practices, much of which is generated by large scale industrial operations. Forest management research in the region was first initiated by FAO in the 1950s (Oliveira 2000), but it was not until the early 1990s that the Brazilian federal government began advocating reduced-impact logging (RIL) techniques as a means to combat rampant predatory logging in the Amazon basin (Holmes et al. 2002, Schulze 2003). RIL is an essential component of SFM if timber is to be harvested from the forest (Pearce et al. 2003), typically employing a pre-harvest forest inventory, road and skid trail planning, pre-harvest vine cutting, and directional felling (Dykstra and Heinrich 1996). Indeed, in the absence of secure prices for non-timber forest products (NTFPs) and infrastructural support for ecotourism efforts, selective logging under the RIL model at low intensities may be one of the few assured income-generating activities

that can preserve the structure of the forest without compromising the entire ecosystem (Oliveira 2000).

The SFM model has generated considerable interest for community managers and smallholders in the region, with the last few years witnessing the creation of a number of innovative community-based conservation experiments founded on SFM principles (Kainer et al. 2003, Stone 2003). Extractive reserves, or tracts of forest under community control with secure usufruct rights, figure prominently among these community development efforts. The remarkable and often violent history that preceded the emergence of such initiatives as the extractive reserve (RESEX) program immediately gave these forest communities widespread international support, but their conservation goals have since become the focus of intense scrutiny (Nepstad et al. 1992, Redford and Stearman 1993, Terborgh 1999, 2000). Many researchers, although generally not contesting the social justice value that such endeavors provide to a historically disenfranchised population, have questioned the wisdom in holding up the extractive reserves as models for biodiversity conservation (Redford and Stearman 1993, Terborgh 2000). This issue has become especially pertinent now that many community forests have shifted from local economies based on non-timber forest products (NTFPs) to that of timber extraction (Kainer et al. 2003, Schmink 2004). The general consensus among some researchers is that external economic and cultural pressures, in concert with a tendency to abuse the resource commons (see Hardin 1968), will hinder the ability of community forest managers to protect biodiversity (Redford and Stearman 1993, Terborgh 2000).

In Brazil, though, voluntary forest certification has provided a strong incentive for both community forest managers and commercial logging companies to adopt SFM practices (May 2002, Kainer et al. 2003). As of April 2004, Brazil ranked fourth in the world in number of certified forests and eighth in area (slightly under 1.6 million ha, 529,079 ha of which are considered natural forest) (May 2002). The failure of government policies to curb illegal logging gave birth in the 1990s to the certification instrument, which endorses forest products originating from sustainable forestry operations, a process often executed by a third-party organization (Molnar 2003). Forest certification's role as a community-based conservation mechanism has received widespread support (Molnar 2003). This has been particularly true of the Forest Stewardship Council (FSC), an international organization that has had a strong influence in the development of certification standards in the Brazilian Amazon (Humphries 2005). Ultimately, the certification process could provide access not only to the global certified forest product market but also to silvicultural evaluation and assistance, thereby creating motivation to conserve the resource base (Putz 1996, Putz and Viana 1996, Putz and Romero 2001).

Despite their positive associations with the conservation movement, a number of researchers have criticized forest certification bodies and biologists for producing SFM guidelines and criteria that fail to recognize the problems that such a system might create for smallholders (Pretty and Hine 1999, Molnar 2003). Even though community timber management operators may have the knowledge, "human nature and economics" can offset the desire to sustainably manage the forest (Salafsky et al. 1998). Smallholders and community forest managers in the rural tropics have limited financial and technical

resources, so implementing additional silvicultural treatments and monitoring may be unfeasible for them, unless the proposed activities are seen as adding to their productivity. Furthermore, given intense competition from less sustainable commercial industrial timber enterprises and illegal logging, the long-term impact that certification will have on the industry and deforestation in general is questionable, particularly in Latin America (Sierra 2001). In Brazil, for example, certified wood is currently meeting only 2% of the annual demand for logs (Lentini et al. 2003).

Community and smallholder forest enterprises are generally thought to be more environmentally benign than their industrial counterparts (Salafsky et al. 1998), making them seemingly ideal candidates to implement the SFM and RIL models. There are, however, a number of inherent ecological limitations associated with small-scale timber operations that are rarely addressed in the management literature, with few ecological investigations regarding the impact of timber extraction having been conducted at the level of the community or smallholder level (Oliveira 2000, Walters 2005). Although RIL is, in general, ecologically beneficial to the residual stand, there is evidence that the magnitude of those benefits varies, depending on the forest being exploited (Fredericksen and Putz 2003, Schulze 2003) and the specific RIL techniques being used. RIL was initially developed for large scale industrial operations where harvest intensities and investment potential are high (Oliveira et al. 1998). Nevertheless, community forest managers have eagerly adopted RIL standards, even though the ecological impacts of RIL at this scale are not well-known. Given that these impacts are poorly understood, there is a need to explore both the limitations and the benefits of RIL for these

community and smallholder-based systems rather than embracing what is held to be the ideal by researchers working with larger operations.

In this thesis I examine both RIL guidelines as generally implemented at the community and smallholder levels as well as a specific case study in PAE Porto Dias, Acre, Brazil, testing the harvesting impacts generated by the activities of the "Multiple-Use Community Forest Management Project" on the residual stand. This approach potentially addresses some of the challenges of community timber operations in Acre as well as examining future crop tree (FCT) damage as sustained during RIL operations.

In Chapter Two, I examine some of the ecological constraints associated with timber operations at the community and smallholder scales. To do so, I evaluate several assumptions of selected RIL recommendations and their implications for smallholder management. After a brief introduction to SFM and RIL, particularly as they relate to the Brazilian Amazon, I discuss eleven key differences between industrial and community/smallholder operations. I then ask to what extent the RIL model contributes to SFM at the smallholder/community level, with an examination of RIL harvesting techniques and their potential influences on the population dynamics of tropical tree species in smallholder systems, with a focus on the Brazilian Amazon. I conclude the paper with a discussion of potential future management strategies for smallholders.

In Chapter Three, I present research investigating the impacts that forest type (bamboo/non-bamboo), liana presence, harvesting intensity, and marking have on FCT damage during logging operations in PAE Porto Dias. The study focuses on the adult and sub-adult size classes (≥ 20 cm dbh) of commercial timber species in the hopes of

capturing information about the availability of timber at the time of the next harvest (30 years).

This thesis has been organized such that the second and third chapters are two individually and fully structured papers for publication. Each one of these chapters has its own conclusions, while Chapter 4 summarizes the main findings of the entire study and provides a discussion of the future directions for small-scale timber management research.

CHAPTER 2
ECOLOGICAL LIMITATIONS OF REDUCED IMPACT LOGGING AT COMMUNITY AND SMALLHOLDER SCALES

Introduction

Researchers and practitioners alike have proposed that large tracts of tropical forests will not be conserved without the involvement of local forest residents (Scherr et al. 2002, Shanley and Gaia 2002). Already, approximately one quarter of the world's forests are community-owned or managed, a proportion that is expected to double in the next 15 years (Molnar 2003). Some have advised that these forests will stand a greater chance in surviving when placed under the jurisdiction of communities, assuming they will have a desire to conserve their forest resource base (Camino and Alfaros 2000). In concert with a growing smallholder interest in formal timber production and a need for sustainable forest management (SFM) practices outside of protected areas (Putz et al. 2000), it is likely that community and smallholder forests will play an increasingly important role in global conservation.

For various reasons, including an international market demand for sustainably harvested timber products, many of these community and smallholder operations are becoming third-party certified (Molnar 2003). This voluntary process emphasizes rigorous social and environmental standards for production forests, potentially providing a strong economic incentive for local forest managers to value the resource base (Putz 1996, Putz and Viana 1996). The reduced impact logging (RIL) management models favored by the certification instrument, however, have failed to develop situation-

appropriate strategies that accommodate the ecological and social complexities of small-scale commercial timber extraction. Instead, the tendency has been to impose management practices of large commercial operations onto smallholder systems (Oliveira 2000), causing some forest operations to take on corporate features usually relegated to industrial timber enterprises (Putz 2004). It has been suggested that further research and operational training are needed to evaluate how variations in forest type and size of logging operations affect the performance of RIL systems (Holmes et al. 2002).

Without much empirical information suggesting otherwise, it is often assumed that because many smallholders harvest at low intensities and utilize animal traction (Oliveira 2000) or small machines, they are more environmentally benign than industrial logging enterprises (Salafsky et al. 1998). Nevertheless, there are a number of inherent ecological limitations associated with small-scale timber operations and low intensity harvests, which have been shown to have substantial impacts on forest structure, composition and regeneration (Smiet 1992, Gullison and Hardner 1993, Boot and Gullison 1995, Whitman et al. 1997, Awasthi et al. 2003, Walters 2005). Community and smallholder-owned forests generally encompass smaller areas compared to their industrial counterparts (≤ 500 ha) (Lentini et al. 2003). When managing for timber in small, non-contiguous management units, ecological constraints to regeneration, such as insufficient pollen transfer and/or inbreeding (Ghazoul et al. 1998), dispersal limitation (Dalling et al. 2002), and predator satiation (Schupp 1992) can become exacerbated, as population sizes of all species are correspondingly small (Putz et al. 2000).

RIL's emphasis on environmental sustainability is often assumed to be a natural fit for rural forest dwellers (Braz and Oliveira 1996). However, it is becoming increasingly

clear that the shift from traditional management systems to that of third-party certified commercial timber extraction is challenging. Smallholders and communities are typically required to implement RIL techniques to meet certification standards, compelling them to invest in both equipment and human capital not normally required for their daily subsistence needs (Stone 2003).

Although the RIL model has been demonstrated to have many ecological benefits in tropical forests, there is a need to explore both the limitations and the benefits of RIL specifically for smallholder systems. In this paper, I evaluate key assumptions of several RIL recommendations and their implications for smallholder management. I use the terms “smallholder” and “community” interchangeably throughout, though there is a distinction (albeit broad) between smallholders living on individual lots in colonization projects and traditional or indigenous forest-dwelling communities, such as those living in or near forests under some form of protection. As well, when referring to the terms “community” and “smallholder” management in this paper, I am specifically addressing locally-run operations, not to be confused with commercial logging concessions on smallholder properties (e.g. Nepstad et al. 2004). After a brief introduction to SFM and RIL, I discuss eleven key differences between industrial and community/smallholder operations. I then ask to what extent the RIL model ensures SFM at the smallholder/community level, with an examination of RIL harvesting techniques and their potential influences on the population dynamics of tropical tree species in small scale systems, with a focus on the Brazilian Amazon. I conclude the paper with a discussion of potential future management strategies for the smallholder system.

Sustainable Forest Management (SFM) and Reduced-Impact Logging (RIL) Models

Selective logging when applied on a long rotational basis and at low harvest intensities is generally regarded to have a low impact on the residual stand (Uhl and Buschbacher 1988, but see Sist et al. 2003a,b, Asner et al. 2005). Selective logging, however, can be one of the first steps in forest conversion, providing a subsidy for pasture formation as in the Brazilian Amazon case (Oliveira 2000). Additionally, very few commercial timber operations in this region employ sustainable selective logging techniques, with an estimated 95% of all timber extracted through unplanned conventional logging (Verissimo et al. 2002).

The extent to which selective logging causes permanent damage can be controlled in some ways by planned management activities, as has been suggested by the SFM model. SFM can be defined based on the following assumptions: (1) management can be exercised in a manner compatible with the maintenance of biodiversity, (2) management of tropical forests is economically viable, and (3) management can result in sustained timber yield over the long term (Bawa and Seidler 1998). Translating these concepts into practice, Bruenig (1996) maintains that when discussing timber extraction, SFM should encompass the following: “canopy openings should be kept within the limits of natural gap formation; stand and soil damage must be minimized; felling cycles must be sufficiently long and tree marking so designed that a selection forestry canopy structure and a self regulating stand table are maintained without, or with very little, silvicultural manipulation; (and) production of timber should aim for high quality and versatility”. Bawa and Seidler (1998) conclude, however, that too often, SFM is promoted without rigorous examination of the interrelatedness between biodiversity maintenance, economic viability, and long-term yields, causing some researchers to doubt the validity of SFM

and calling for the outright protection of forested regions instead (see Rice et al. 1997, Bowles et al. 1998, Terborgh 1999). As well, some researchers have questioned whether timber harvests in at least some forests can be sustained without silvicultural intervention (Daube et al. 2005). Nevertheless, the SFM philosophy has been embraced by multiple forest actors. In Brazil, for example, the federal government, state institutions, NGOs and research organizations have actively promoted SFM, hoping that it will quell unchecked land conversion and predatory logging activities (Schwartzman et al. 2000, Nepstad et al. 2002, Fearnside 2003, Kainer et al. 2003).

RIL is considered a necessary component to achieve the goals of SFM, though its implementation alone cannot maintain biodiversity and ecosystem functions nor even secure sustained timber yield (referring to the objective of maintaining timber production for future harvests) (Webb 1997, Putz et al. 2000, Sist et al. 2003b). RIL implies a series of pre- and post-logging guidelines (see Pinard et al. 1995, Dykstra and Heinrich 1996) which are designed to protect advanced regeneration (i.e., seedlings, saplings and larger future crop trees), minimize soil damage, limit damage to non-target populations (i.e.wildlife and non-timber plants species), protect water quality, and preserve ecosystem processes (i.e. carbon sequestration) (Putz et al. 2000). Operations under these guidelines typically employ long-term cutting cycles, pre-harvest forest inventories, road and skid trail planning, pre-harvest vine cutting, and directional felling (Dykstra and Heinrich 1996). RIL has been shown to reduce damage to the residual stand by 30-50% and has also demonstrated an increased efficiency of logging operations (see Johns et al. 1996, Pinard and Putz 1996, Bertault and Sist 1997, Barreto et al. 1998, Sist et al. 1998, Holmes et al. 2002).

Under pressure from both domestic and international concerns to improve the ecological sustainability of the tropical hardwood industry, Brazil began to adopt RIL techniques in the early 1990s, following the model that was originally developed in Southeast Asia and Suriname a decade earlier. Since that time, many Brazilian timber companies have agreed to abide by RIL regulations at the behest of federal government regulations, and in some cases, third-party forest certification standards, hoping to tap a high-end niche market (Schulze 2003).

Community/Smallholder Scale vs. Industrial Operations

The SFM model (and implicitly, RIL) has generated considerable interest for community forest managers and smallholders (Braz and Oliveria 1996). Researchers often point to local peoples' "long-term commitment to place" and their intimate knowledge of ecosystems (Schmink 2004) as evidence of their ability to both manage and (when necessary) restrict access to forest resources (Agrawal and Gibson 1999). Yet, it has also been noted that the implementation of a systematic management regime can often be overwhelming for communities and smallholders, requiring significantly different conditions for long-term success than large-scale commercial enterprises (Schmink 2004). Indeed, to mitigate the challenges faced by these operations, some suggest that community and smallholder forest management systems must be considered within case-specific ecological, socio-economic, political, and cultural settings (Berkes and Folke 2001, Schmink 2004). This viewpoint recognizes the unique character of individual community and smallholder systems and also further emphasizes the need to distinguish these enterprises from their industrial counterparts.

Several general operational characteristics distinguish community and smallholder timber enterprises from the large-scale commercial timber industry, building on a list of

five proposed by Salafsky et al. (1998): (1) **Ownership:** Industrial operations usually buy logging rights via a concession, or the timber company may own the land. More than likely, the proprietors live offsite. In contrast, community and smallholder timber harvesting rights are owned or, at the very least, held by local inhabitants (Salafsky et al. 1998). For example, in the Brazilian Amazon, one third of the land area is inhabited by smallholder and indigenous communities with the rights to manage their own forests (Shanley and Gaia 2002). Due to a strong sense of ownership and multigenerational responsibilities, community forest managers may implement more sustainable management practices than their industrial counterparts (Walters et al. 2005). Indeed, in those community cases where secure land tenure is lacking, there appear to be fewer incentives to manage the resource sustainably (Banana and Bombya-Ssembajjwe 2000, Gibson and Becker 2000). **(2) Timber as integrated livelihood component:** Community and smallholder forest managers rely on the forest for a suite of goods and services, viewing timber management as one component of an integrated system rather than a sole income-generating activity (Salafsky et al. 1998). Since timber harvesting is a seasonal activity (usually during the driest months), local managers pursue other livelihood activities throughout the remainder of the year (Oliveira 2000). Even so, timber management and other livelihood activities are not always compatible, especially when harvesting is conducted by those from outside the community. In one Eastern Amazonian case, community members from the Tapajós-Arapiuns Extractive Reserve believed that commercial logging negatively impacted NTFP collection (due to incidental damage) and hunting (through easier game access by logging roads) (Menton 2003). Researchers also have reported a lack of documented systems explicitly integrating timber and NTFPs in

management plans (Bawa and Seidler 1998, but see Salick et al. 1995, Romero 1999). Local forest managers who wish to protect a multitude of forest services (NTFPs, game species, watershed) may be more inclined to adopt RIL guidelines geared to protect components of the forest beyond future timber harvests. (3) **Scale:** Community and smallholder enterprises generally operate on a smaller scale than their industrial counterparts, the latter tending to rely on capital intensive techniques and heavy machinery rather than local labor (Salafsky et al. 1998). Many techniques developed specifically for communities have been initiated in secondary forests, where the use of animal traction is ideal for the extraction of small diameter logs (Pinedo-Vasquez et al. 2001). Smallholder logging in primary forests has become more common in recent years, particularly in the Brazilian state of Acre (see Kainer et al. 2003, Stone 2003), but the need for heavy machinery in these systems and the subsequent competition with larger enterprises has constrained community participation in this type of extraction (Oliveira 2000). However, smaller scale operations may facilitate more intensive management for a suite of services and products, something that might be impractical in a larger management unit. Also, in Brazilian community timber efforts, harvest intensities tend to be low, usually no more than 10 m^3 (or two trees) ha^{-1} (Oliveira et al. 1998), which (in theory) should be compatible with RIL guidelines. In comparison, most Brazilian conventional forest management systems harvest anywhere from 30 to 40 m^3 ha^{-1} yr^{-1} (Oliveira 2000). (4) **Added-value:** Industrial logging operations harvest roundwood logs that are then transported to a centralized offsite-processing mill. Conversely, community and smallholder operations often seek to add value to the lumber onsite, transforming the raw material into finished or semi-finished products (Salafsky et al. 1998). This

potentially provides greater income to local residents, such as the case of at least one certified community operations in Acre, Brazil that is producing artisan objects from locally harvested wood (*pers. obs.*). (5) **Local reinvestment:** In community-based systems, capital is generally reinvested locally and there is a greater long-term incentive for sustainability (Salafsky et al. 1998). In comparison, industrial timber enterprises tend to move profits to other localities or sectors of the economy, thereby reducing any potential social benefits from forestry activities (Salafsky et al. 1998). For example, in Oaxaca, Mexico, Bray et al. (2003) note community use of logging profits to encourage more benign forest enterprises (i.e., water bottling, ecotourism, and resin-tapping) and build more community assets (i.e., potable-water networks, schools, and clinics). And even though timber extraction has subsidized cattle pasture expansion in the Brazilian Amazon (Oliveira 2000), Summers et al. (2004) found that smallholder timber managers in Rondônia tended to clear less forest and had smaller cattle herds than their neighbors. (6) **Adaptive management/traditional governance:** Communities and smallholders sometimes develop adaptive management systems that permit local users to identify internal and external threats to the forest (Margoluis and Salafsky 1998, Salafsky et al. 1998). Some researchers have recognized this as a benefit of traditional governance, or locally-defined institutions and regulations that guide decision making processes (Banana and Gombya-Ssembajjwe 2000, Becker and León 2000, Bray et al. 2003). For instance, Becker and León (2000) found that the Yuracaré of the Bolivian Amazon have clear, locally-defined rules regarding forest resources, upon which they can control both access and use of the forest at multiple scales. (7) **Value of forest to future generations:** Intergenerational values or investment in the forest for future generations also distinguish

community timber enterprises from their industrial counterparts, a factor often overlooked (see Bowles et al. 1998) when discussing excessive production costs of community forests (Bray et al. 2003). In several studies conducted in the Brazilian Amazon, rural respondents identified silvicultural activities as positively impacting the future of their children and grandchildren (Walters et al. 2005). (8) **Traditional ecological knowledge:** Rural peoples who have often lived in the same geographic region for generations have accumulated considerable knowledge about the natural resource base through daily observations in their subsistence activities, usually demonstrating a comprehensive understanding of ecosystem linkages (Folke et al.1998, Berkes 1999, Calheiros et al. 2000, Berkes and Folke 2002). For example, it has been noted that despite the highly technical silvicultural expertise available in Finland, the main factor explaining the excellent forest health in that country is that conservation practices have been based on TEK passed from one generation of forest managers to another (Banuri and Apffel-Marglin 1993). Accordingly, TEK can be applied to a systematic forest resource management plan, providing details on timber quantity and location, regeneration requirements, and other requisites for successful implementation of RIL guidelines. (9) **Commitment to forest conservation:** Mexican communities in Oaxaca and Quintana Roo consistently logged below the authorized volume (some by as much as 37%), suggesting a willingness to reduce the volume of extraction when inventories indicate external threats to the resource base. Some of these same communities also prohibit or limit hunting in local forests, displaying an increasing commitment to biodiversity conservation (Bray et al. 2003) or perhaps, a clear understanding of sustainable game harvest limits. Other forest managers in Acre, Brazil

have developed local indicators for advanced regeneration as determinants for harvest. By identifying a sufficient number of what they term as "daughters" and "granddaughters" of the tree(s) to be harvested, they acknowledge the importance of leaving several individuals of differing size classes of a particular species in the management unit, not just reproductive adults (P. Roth, *pers. comm.*). (10) **Participatory monitoring:** Local participants, especially those termed "parabiologists" or "parataxonomists", have been instrumental in monitoring environmental impacts and planning formal conservation initiatives, and, in some cases, have actually improved the data quality of the academically-trained researcher (Janzen 2004, Sheil and Lawrence 2004). In one Brazilian case, local participants monitored the populations and phenology of selected fruit tree species in collaboration with professional biologists, eventually determining that the environmental impact of timber extraction was too great a risk to the long-term NTFP value of a particular fruit species (Shanley and Gaia 2000). These examples suggest that local forest dwellers are very aware of forest dynamics and capable of executing environmental monitoring (especially those post-logging surveys encouraged by the RIL paradigm) when provided with some capacity building and equipment (Sheil and Lawrence 2004). (11) **Influence/funding from outside organizations:** Particularly since the early 1990s, a network of community forests in the Brazilian Amazon have received financial and infrastructural support of a number of NGOs and governmental institutions (Stone 2003). This support has led to the voluntary third-party certification of at least 7 community operations in the region, fortifying the competitiveness of community forest management in a demanding market dominated by industrial giants. Indeed, two of the principal barriers faced by community and

smallholder managers are the small scale production and consequent marketing difficulties, limitations that certification may be able to improve, primarily through access to a niche market (May 2002). Yet, Stone (2003) also warns of the increasing reliance of certified community timber projects on outside funding and technical assistance, mostly because of the complex requirements inherent with RIL adoption, such as investment in expensive equipment and the writing of management plans.

To What Extent Does the RIL Model Ensure SFM at the Smallholder/Community Level?

Prior to the recent emphasis on the RIL model, traditional smallholder forest exploitation in the Brazilian Amazon was based on low technology methods that produced minimal financial returns, but also resulted in low environmental impacts (Oliveira 2000). Indigenous inhabitants have been managing forests in the region since pre-Columbian times (see Heckenberger et al. 2003) for a suite of subsistence products and services (Summers et al. 2004), a pattern that has continued to the present. Rural Amazonian inhabitants typically integrate wood production into their existing swidden-fallow systems, resulting in a highly patchy landscape and allowing smallholders to maintain favorable environments for several different species (Pinedo-Vasquez et al. 2001). It was not until the early 1990s, amidst a deepening international concern for global deforestation and the creation of the Program for the Conservation of the Brazilian Rain Forest (PPG7) in 1992, that the formal establishment of SFM projects for community and smallholder forests emerged (Stone 2003), eventually leading to the first RIL project for privately-owned smallholder forests in the state of Pará in 1993 (Amaral and Neto 2000). By the mid 1990s, the Brazilian federal government also began to take an interest in community-based SFM models and in 1998, the Brazilian Institute of the

Environment and Renewable Resources (IBAMA) (the federal agency directly responsible for the execution of federal forest policy) implemented the PMFSimples (Simple Sustainable Forest Management Plan) protocol, allowing smallholders to manage up to 500 ha for timber (Stone 2003). It also enabled community forest initiatives to submit one management plan for all participating properties, rather than do this individually. Since 1995, at least 7 smallholder and community forest projects have been established in the region, under a wide range of property regimes (extractive reserves, national and state forests, indigenous reserves, and agricultural colonization projects) (Stone 2003). Community and smallholder interest in timber has been motivated in great part by a need to increase and diversify livelihoods, fueled in particular by low net returns for NTFPs (Kainer et al. 2003).

However, as the sophistication and intensity of community and smallholder forest management systems increases, it has also become evident that silvicultural prescriptions appropriate to local conditions are needed, especially those that take into account a shortage of investment and human labor (Oliveira 2000). The Brazilian RIL model was initially developed for large scale industrial operations where investment potential is high (Oliveira et al. 1998). For example, since 1995, training in RIL techniques for Brazilian operations has been a primary focus of the Tropical Forest Foundation (TFF) and its Brazilian subsidiary, Fundação Floresta Tropical (FFT) (Holmes et al. 2002). This effort has resulted in the development of several model systems throughout the Amazon basin, but most of these examples are considered large-scale operations. Not surprisingly, these types of models have been eagerly adopted by community forest managers, even though

the RIL ecological impacts on the overall forest structure at this scale are not well-known (Oliveira et al. 1998).

In its original conception, one of the basic tenets of RIL mandated that logging should mimic the so-called gap-phase dynamics of the neotropical rainforest (see Brokaw 1982) by creating small-scale disturbances around the extracted crop tree (Hartshorn 1989). Natural small-scale disturbances (usually involving the death and subsequent fall of a neighboring dominant tree) permit access to limiting light (and other) resources. By attempting to mimic this type of disturbance during RIL operations, it is hoped that residual stand damage will decrease and that growth rates of surviving trees will increase. Natural disturbances, however, are qualitatively and quantitatively different from those logging-induced disturbances (Boot and Gullison 1995, Vitt et al. 1998, Dickinson et al. 2000). Boot and Gullison (1995) note that mortality rates of trees in forests with only natural disturbances are on the order of 1-2% (see Swaine et al. 1987), while conventional selective timber harvests can damage as much as 55% of the residual stems. Additionally, they point to other sources of residual stand degradation, such as soil compaction, erosion, and changes in drainage, that are not part of a natural small-scale disturbance.

Although RIL techniques are designed to minimize this type of aggravated impact, some species of commercially-valuable trees require larger disturbances for reproduction than is mandated by minimal impact logging techniques, which usually emphasize low harvest intensities (Fredericksen and Putz 2003). Careful planning may maintain timber volume but the removal of scarce, high value logs could result in replacement by less important species, ultimately reducing forest biodiversity and dooming the second cutting

cycle to timber not yet recognized by the market (Schulze 2003). In this case, the gap-phase management paradigm could be in direct opposition to the ecological role of catastrophic natural and anthropogenic disturbances in forests, conditions to which particular species have adapted over time (Deneven 1992, Bush and Colinvaux 1994, Fredericksen and Putz 2003). Fredericksen and Putz (2003) point to examples of neotropical trees that may not benefit from a low-impact management regime, such as *Swietenia macrophylla*, *Bertholletia excelsa* and *Cedrela spp.*, species which appear to establish in large disturbed areas.

Another of the underlying assumptions of many RIL guidelines is that advanced regeneration of valuable commercial timber species is abundant and exists uniformly throughout the stand (Schulze 2003). However, given the suite of differing growth rates, wood densities, population dynamics, and modes of seed dispersal, both between and within commercial species in the Amazon basin (Schulze 2003), it may be that knowledge of tree autecology is one of the greatest limiting factors in implementing SFM (Hammond et al. 1996, Guariguata and Pinard 1998). Currently, there is a lack of sufficient ecological information to help guide forest management decisions in the Brazilian Amazon, which is one of the reasons forestry methods have not advanced much beyond RIL (Schulze 2003). Yet, it is prudent to evaluate small-scale, RIL-based logging operations to date, and discuss how some of the basic tenets of RIL (pre-harvest forest inventories, selection of harvest and seed trees, minimum diameter cuts, low harvest intensity, skid trail planning, rotation cycles, and liana cutting) have worked at the smallholder level and how they can be modified.

Pre-harvest Forest Inventories, Botanical Identification, and Selection of Seed Trees

A full pre-harvest inventory provides managers with detailed information on distribution, quantity, and size of potential harvest trees and future crop trees (Schulze 2003). Unfortunately, for many small operations (as well as some large ones), 100% inventories are cost prohibitive, especially when the entire management unit (as is the case of those community management projects in the extractive settlement projects and extractive reserves in Amazonia) is as large as 300 ha. In at least one study in Quintana Roo, Mexico, local rural people identified time and finances as two of the biggest constraints for the inventory process (Lawrence and Roman 1996).

In response, it has been common to inventory only the logging block for the following year, as the inventories are generally carried out one year prior to harvesting activities. Although this may eliminate the need to pay for a costly comprehensive inventory, it limits the manager's ability to harvest with respect to species population biology (see Table 2-1). For example, in a 10 ha logging block chosen from a 300 ha landholding, several individuals of a particularly valuable species are identified during the 10 ha inventory. For management purposes, these individuals are then the sole representatives of that species within the 300 ha landholding, since a full inventory of adult trees has not been carried out. Many commercial tropical tree species are characterized by highly clumped spatial distributions (Hubbell 1979; Table 2-2) and are self-incompatible (Bawa et al. 1985), so harvesting even half of the inventoried individuals could potentially lead to inbreeding and genetic drift (Jennings et al. 2001).

Ghazoul et al. (1998) demonstrated that the reproductive success of *Shorea siamensis* was limited by compatible pollen transfer at a reduced population density after logging. This Allee effect (see Allee 1949) can increase the probability of extinction,

particularly in small populations. For those species occurring in widely spaced clumps, a reduced density in aggregations may result in a decrease in pollen exchange between clumps, severely limiting genetic variability within aggregations and thus, their ability to adapt to changing conditions (Ghazoul et al. 1998).

When asked about this type of problem, local forest managers often respond that they know their forest well enough to have a general understanding of species distribution patterns (*pers. obs.*). This might be true in many cases, but at least one study conducted in Pará, Brazil revealed that during Rapid Rural Appraisal interviews, local residents grossly overestimated the densities and distributions of game-attracting fruit species, presumably because they were favoring forest groves where the tree species occurred in high densities (Shanley and Gaia 2002). Even if the landowner or community cannot afford to pay for a 100% inventory, other forms of surveys (like rapid-assessment type inventories) should be tested to give the forest manager a more accurate idea of the availability of species and their distributions across the landholding. For example, in the *ejido* system of Quintana Roo, Mexico, Lawrence and Roman (1996) report that local inventory specialists prefer small circular plots as opposed to a 100% inventory design. Although these circular plots may lack some of the statistical accuracy of a full-scale inventory, they satisfy local needs for ease of implementation and available financial resources.

Botanical identification during the inventory is crucial for making the most basic management decisions and should be considered one of the first steps in SFM. In Brazilian community forest settings, much of the identification is carried out by a forest manager or tree identifier (*mateiro*) from the community. Usually in these cases, initial

identifications are recorded as common names. Thanks to the substantial botanical knowledge base in the community, most *mateiros* are able to identify species to common name with few difficulties, using a series of morphological features (i.e., bark, slash, and odor). The greater problem arises when the inventory sheet is sent back to the office of a local NGO or state agency and entered into a database. At this point, the common names are converted to scientific epithets based on lists (often created by a state or federal agency distant from the management region) that have been circulated between agencies. Too often, these lists do not correspond correctly to all common names, as the master list has been generated by a different set of *mateiros*. The assumption that these common names will then correspond with each other is highly optimistic, as even *mateiros* in the same community will often use different names for the same species. In Lacerda and Griffiths' (2005) study from the Tapajós forest in Pará, for example, more than 90% of common names were technically correct, but once the inventory sheet arrived in the office and was converted to corresponding scientific names, botanical accuracy was reduced to less than 40% correct. This type of error can have serious repercussions for both the selection of harvest trees and post-harvest population dynamics.

The pre-harvest inventory and subsequent botanical identification have strong implications for the selection of seed trees. Some RIL guidelines require that "mother trees", or seed trees, are maintained in the management unit at the "appropriate" spacing and density, based on means of dispersal, individual size, and tree diameter at first reproduction, as suggested by Guariguata and Pinard (1998). The seed tree concept does not elude forest managers on the ground. One landowner in a community timber management project in Acre, Brazil expressed his reluctance at cutting down large-

crowned individuals of *Dipteryx spp.* (even though such trees would bring an excellent price on the market), believing them to be good seed sources (as well as being concerned about the subsequent large gap size created when felled). He would instead take a smaller conspecific, or what is locally known as a "daughter" tree (F. Correa da Cunha, *pers. comm.*).

Target mother tree densities are often around 3-4 trees ha^{-1}, or (as in accordance with federal Brazilian law) 10% of the harvest compartment, but given that reproductively mature conspecific densities in many neotropical forests are much lower (Lieberman and Lieberman 1994, Guariguata and Pinard 1998), these target goals may be inadequate. As well, if individuals cannot be properly identified to species, their selection as seed trees remains dubious. For example, current community timber operations in Acre identify *Dipteryx spp.* to one common name, *cumaru ferro*, even though at least 7 species of this genus exist in the region (C. Baraloto, *unpublished data*). Consequently, in a hypothetical situation where 30 mature *cumaru ferro* in a 10 ha logging block have been left as seed trees, it is highly probable that this group entails several species that may be isolated from other conspecific reproductive adults.

In general, the dilemmas of implementing a pre-harvest inventory, identifying scientific species, and selecting seed trees can occur at both the community and industrial scales. Brazilian federal law requires all companies to survey their logging concessions prior to harvest, regardless if they are certified or not. The difference for community and smallholder operations is that a pre-harvest botanical inventory will require long-term budgetary commitments not normally required on the part of a community-based organization or forest-dwelling individuals (Janzen 2004). Local parataxonomists, during

the time of the inventories, are taken away from other responsibilities with families and subsistence activities, potentially compromising their situations at home. As such, they may need to be compensated financially by the forest management group, which can cause resentment among untrained community members who do not benefit directly from the botanical survey.

Minimum Diameter Cuts

The most common harvesting method in the tropics is selective logging, in which only the largest, most valuable individuals are removed from the forest (Webb 1997). One of the justifications used to support the practice of employing a minimum diameter cut is that smaller individuals of the same species will grow to replace the harvested trees. However, application of minimum-diameter cuts (often termed "high grading", or taking the biggest and the best) is riddled with undesirable outcomes, and has thus been criticized as a "short-term gain for long-term loss" (Nyland 2002). As minimum diameter cuts are often defined by market demands rather than tree biology, Sist et al. (2003a) suggest that diameter limits be based instead on the structure, density and diameter at reproduction of target species. Minimum diameter cuts are not necessarily an inherent part of RIL, but their implementation is linked to the polycyclic cutting cycle, which is a common RIL recommendation.

High grading typically dooms the next harvest cycle to small, poorly-formed trees of lesser commercial value or, at the very least, leaves inferior genetic stock to produce the next generation (Smith et al. 1997, Fredericksen and Putz 2003). For example, Vester and Navarro-Martínez (2005) point out that selective logging of the largest trees may deplete a population of the most favorable genotypes, eventually leading to genetic erosion. The extent to which this selection will affect populations depends on a number

of factors, including harvest intensity, the contribution of the residual trees to regeneration, and the degree to which undesirable traits are genotypic rather than phenotypic (Jennings et al. 2001). As well, strong correlations have been drawn between tree size and fecundity (Chapman et al. 1992), so the removal of a stand's largest individuals often eliminates the most prolific seed producers, important for both wildlife as well as regeneration (Thiollay 1992).

Most tropical rain forests have more individuals below the minimum diameter cutting limit than above. When the reverse is true, the population is often assumed to be in decline (Jennings et al. 2001). One forest type representing the latter category is the arborescent bamboo (*Guadua spp.*)-dominated forest of southwestern Amazonia, which has been overlooked by the literature for management considerations (see Chapter 3). Griscom (2003) hypothesizes that if *Guadua* achieves dominance in managed forests, either the majority of tree stems will become competitively excluded, or a substantial proportion of the remaining stems will become deformed as a result of mass loading (referring to the added physical mass of the bamboo, which can displace aboveground tree structure (Griscom 2003)). He notes that it is not necessarily the inability of juvenile trees to germinate and establish in *Guadua* stands that distinguishes the reduced above-ground biomass in these forests, but rather the failure of seedlings and saplings to recruit into larger size classes. These factors present a few challenges to the minimum diameter cut, such as lack of advanced regeneration (one of the requisites for RIL) and removal of a substantial portion of what little canopy exists in the form of large trees (*pers. obs.*). As such, the stand dynamics of this particular type of forest might lead to an increased

probability of high grading. Yet many smallholders, because they are limited to their landholdings, may be forced to harvest in these areas.

Another major drawback of the minimum diameter cut affecting both large and small-scale operations is that knowledge of tree size at first reproduction (which is essential for maintaining adequate regeneration levels) is very limited in the neotropics (Guariguata and Pinard 1998). In the Yucatán forests of Mexico, for example, an original 60 cm dbh diameter limit for *Swietenia macrophylla* (later reduced to 55 cm) was based not on ecological constraints, but rather, market demand (Snook 1998). Even with the new 55 cm limit, it is assumed that it will take 75 years for the next generation of *S. macrophylla* to reach that minimum, an assumption that has been met with much skepticism (Vester and Navarro-Martínez 2005).

Brazilian forest law requires that all harvested timber trees must be ≥ 45 cm diameter at breast height (dbh) (Schulze 2003). This is considered to be a fairly conservative lower limit by many forest managers, and some impose their own higher diameter limits on certain species, based on what they think is the ideal sapwood-heartwood ratio of the tree (F. Correa da Cunha, *pers. comm.*). Whereas a few species are characterized by little sapwood (e.g. *Peltogyne spp*.) and thus are considered worthwhile to harvest at diameters as small as 45 cm dbh, many species are not harvested unless they are much larger. In one community operation in Acre, Brazil, *Tabebuia spp.* and *Hymenaea spp.* must reach 70 cm dbh before they are cut, whereas other taxa, such as *Dipteryx spp.*, must attain 80 or 90 cm dbh before they can be considered for harvest. This management strategy is also employed in operations that manage vast forest concessions, and in either case, such a minimum diameter limit could remove many of

the largest trees on the site. Schulze (2003) suggests that by increasing the minimum diameter cut to 60 cm dbh, one runs the risk of removing the most vigorous individuals on site, not necessarily the oldest. In this scenario, it would be best to employ a proportional cutting across adult size classes rather than concentrate efforts on a minimum harvestable size, thereby ensuring the maintenance of fast-growing individuals for the next cutting cycle.

Low Harvest Intensity

Low harvest intensities (few trees harvested per hectare) are common in smallholder systems, and indeed, low intensity selective harvesting is attractive to those conservationists promoting SFM (Whitman et al. 1997). In small community management units in Acre, Brazil, the desire to maintain NTFP resources coupled with local investment in timber certification initiatives have encouraged communities to try and create as little damage as possible to the residual forest during logging operations (*pers. obs.*). This aspiration is also apparently reflected in Oaxaca and Quintana Roo, Mexico, where community forest managers generally log below the authorized volume (Bray et al. 2003).

However, low harvest intensities associated with these smallholder timber operations may actually aggravate logging disturbance. Several studies have documented that as intensity of harvest increased, fewer skid trails and logging roads were required, resulting in a declining per-tree damage rate (Gullison and Hardner 1993, Whitman et al. 1997, Panfil and Gullison 1998). Panfil and Gullison, in their 1998 study of short term impacts of experimental timber harvest intensities in the Chimanes Forest of Bolivia, cautiously argue for an intensive (albeit at a fixed volume), localized harvest rather than an extensive extraction over a larger area. They suggest that the primary benefit of doing

so is the reduction in mortality of smaller size classes due to skid trail construction and use. The tradeoffs involved with this scenario include the following: first, by logging one area of a large management unit more intensely, there is an increased chance of reducing local population levels. Second, even if the area damaged due to skid trail and road construction is decreased, there will be more concentrated disturbance in the treefall area in the form of large gaps, which are more susceptible to small-scale disturbances, such as windthrow and fire (Webb 1997). Third, the threshold values for the release of pioneer and vine species after such harvests is unknown for many forests, necessitating monitoring plots for this potentially site-specific information (Panfil and Gullison 1998). Fourth, the impacts of logging damage should ideally consider biodiversity maintenance in the event of a more intensive and localized extraction (Panfil and Gullison 1998).

Tradeoffs associated with low harvest intensities will necessarily vary, depending on site and operational constraints. One of the primary factors that should be taken into consideration is the location and abundance of commercial species in a given logging block and to compare these values with other management units (if they exist) in the project. If considerable skid trail construction is required to reach one individual of a desired species, the destruction and cost necessary to harvest it may outweigh the benefit of acquiring the log. In this case, it may be possible to take another individual of the same species in a management unit better suited to its harvest (i.e. greater abundance of the targeted species or closer proximity), if there are multiple landowners harvesting simultaneously in one year as a community effort. It should not be assumed that just because an operation is extracting at a low intensity that it is having minimal impact on

the forest. It is possible to have few commercial-sized trees injured during logging while simultaneously incurring substantial ground and smaller stem damage (see Chapter 3).

Skid Trail Planning

One way to minimize extraction damage is to reuse old skid trails, but this method can prove challenging for smallholders, depending on the management system being implemented. For example, in the "Multiple-Use Community Forest Management Project" in the extractive settlement of Porto Dias in Acre, Brazil, harvests are conducted in 10 ha (200 m x 500 m) logging blocks within individual landholdings (approximately 300 ha), with at least five landholdings participating in a given year. The majority of the first logging blocks (from 2000) in the participating landholdings were located near existing roads, which greatly facilitated the removal of stems from the logged areas and the associated construction of log decks. Logging blocks for the subsequent years were then positioned next to these initial logged sites so that old skid trails from previous years were reused. This is beneficial to the forest managers in that they do not need to construct an extra length of primary skid trail (although it does require them to open up these trails again every year), and it is favorable to the forest because the amount of superfluous skid trails is minimized. Using this method, though, means that future tree regeneration will never become permanently established along the length of these reused skid trails. It also requires the constant perturbation of the previous logging blocks.

In comparison, the forest management project of São Luis de Remanso (another extractive settlement project in the state of Acre) utilizes 50 ha logging blocks, slightly reducing the need to reenter the management area in sequential years (depending on the number of members that remain in the project and the yearly execution of management plans). Instead of several different landholdings simultaneously participating in the

harvesting (as is the case in Porto Dias), only one landholding (in general), or 50 ha, is logged per year. In comparison to the Porto Dias system, this setup may be more economically viable, as the forest managers usually harvest in just one landholding per year, so ground efforts are not dispersed. Conversely, in Porto Dias, if five landholdings are logged within a given year, the logging crew is compelled to maintain roads and construct/reopen skid trails in several sites (N. Marcondes, *pers. comm.*), adding extra cost and effort to the operation.

Rotation Cycles

One prerequisite for RIL is the implementation of growth and yield data for different management regimes to guide the choice of rotation cycle (Silva et al. 1995). In the Brazilian Amazon, the rotation cycle generally ranges from 30-35 years, based on an average diameter growth rate of 0.5 cm year $^{-1}$ (Silva et al. 1995). Silva et al. (1995), however, caution that ongoing measurement and analysis of permanent plots over the next 30 years are needed before forest managers embrace an ironclad management prescription. They point to the variable diameter increments in any given class of trees, based in part on environmental conditions, growth habits, and genotypes.

Another drawback of the Brazilian harvest rotation is the application of silvicultural recommendations generated in one region to another. By 1989, approximately 105 ha of permanent plots had been established in the Brazilian Amazon, but most of them are located in the state of Pará (Silva et al. 1995). Meyer and Helfman (1993) point to the need for sustainable resource management plans based on an understanding of specific local conditions. Many of the Brazilian community timber operations are emerging in the western state of Acre (see Kainer et al. 2003), yet these management systems tend to base their diameter and volume increment projections on eastern Amazonian data. One

exception is the permanent plot study that has been established in the Pedro Peixoto Settlement Project by the Brazilian Agricultural Research Corporation (EMBRAPA), which has based the smallholder management system on locally-generated growth and yield data. Drawing from the results generated by the project, Oliveira et al. (1998) propose that extraction of fewer trees per hectare at shorter intervals, in concert with silvicultural treatments, will maintain pre-intervention forest structure and biodiversity levels by creating a gap mosaic of different ages. In other words, a short rotation cycle (in this case, ten years whereby four hectares are harvested annually on a 40-ha stand) will substitute for the larger impact caused by more intensive interventions.

Yet, even if site-specific growth and yield models state otherwise, a short rotation could severely degrade the quality of the resource base. Webb (1997) warns that although increased light levels due to removal of harvested individuals will generally promote the rapid growth of residual trees, repeated harvests over short rotation periods will potentially result in low levels of hardwood regeneration as well as a highly discontinuous canopy. Oliveira (2000), though, counters that damage levels from short interval cycles can be mitigated to a certain extent by reusing old logging roads and skid trails. As well, he makes the observation that the Brazilian SFM system is difficult to apply to the smallholder setting given that the required rotation cycles (20-30 years) make the clearing of forest for agriculture and pasture a more enticing option since the profits from those activities are much more immediate.

Liana Cutting

Proponents of RIL regard the pre-harvest cutting of lianas on crop trees essential if canopy and future crop tree damage are to be minimized and lianas are abundant (Fox 1968, Appanah and Putz 1984, Vidal et al. 1997, Gerwing and Vidal 2002, Pereira Jr. et

al. 2002). On average, lianas connect each canopy tree to anywhere from three to nine others in Amazonian *terra firme* forest (Vidal et al. 1997). During tree felling, this liana connectivity inadvertently can bring down several other trees, creating a far larger gap than would occur if lianas associated with harvest trees were pre-cut (Appanah and Putz 1984, Vidal et al. 1997). This suggests that directional felling (a basic tenet of most RIL guidelines whereby harvest trees are felled in a specified direction to minimize residual stand damage), is less effective when interconnected vines are not eliminated (Fox 1968). Furthermore, as these uncut lianas fall with the crop tree, they typically resprout vigorously, thus competing with new tree recruits and reducing establishment success in gaps (Schnitzer et al. 2000, Gerwing 2001).

Yet, there is little doubt that lianas are essential components of tropical forests, providing an important food source for wildlife species as well as supplying a canopy pathway for some arboreal fauna (Putz et al. 2001). Lianas also contribute to canopy closure after treefall and stabilize the microclimate of the subcanopy layer (Schnitzer and Bongers 2002). When asked about the disadvantages of cutting lianas, local forest managers in Acre, Brazil articulated concern for game species because many liana fruits are available when tree fruits are not (*pers. obs.*), an observation that is corroborated by several scientific studies (Heideman 1989, Morellato and Leitao-Filho 1996). These same forest managers have also observed that lianas constitute pathways between individual tree crowns for many animals (especially primates), something that has also been substantiated in the literature (Schwarzkopf and Rylands 1989, Bobadilla and Ferrari 2000). Consequentially, local logging crews from some Brazilian community forests have expressed reluctance at cutting lianas prior to the felling cycle (*pers. obs.*).

Implementing this particular RIL recommendation might present a few quandaries for the community or smallholder forest manager who is looking to maintain more than just the next cycle's timber yield, a consideration that may not manifest itself with an industrial commercial logging operation. Yet, not cutting lianas presents an obvious tradeoff, as they negatively influence growth, leaf production and sexual reproduction of host trees (Parren 2003). This is an important consideration, particularly in a landholding that is limited in size, and, consequently, in future crop tree abundance. Perhaps more importantly, though, is the shear risk that the logging crew takes when felling a tree that has not been freed of climbing vines (Amaral et al. 1998), a factor that is valid at both the smallholder and industrial scales. Parren and Bongers (2001) propose that the benefits of cutting or not cutting lianas prior to logging is heavily site dependent, so it may be that local forest managers must experiment with this facet of the RIL paradigm before making a final decision as to pre-harvest silvicultural prescriptions.

Conclusions

In this review, I have attempted to highlight some of the ecological constraints of RIL as practiced at the community and smallholder scales, mostly drawing upon examples from the Brazilian Amazon region. Many of the limitations outlined here can also be applied to the industrial commercial setting, such as the importance of a pre-harvest botanical inventory. However, a key distinction with smallholders is that these forest managers often extract timber from undersized management units, necessarily reducing the size and often the density of the populations from which they are harvesting. Ghazoul et al. (1998) point out that SFM (and presumably RIL) requires the maintenance of viable tree populations, but admit that minimum effective population sizes and

densities are rarely known. Logging, even under the RIL model, can reduce population viability when conspecifics are isolated from each other (Ghazoul et al. 1998).

Although RIL is, in general, ecologically beneficial to the residual stand, there is evidence that the magnitude of benefits varies, depending on which forest system is being exploited (Fredericksen and Putz 2003, Schulze 2003). Some logging guidelines outlined in this review are unlikely to change in the near future, simply because they are part of Brazilian federal law, such as the requisite rotation cycle and the minimum diameter cut. Nor perhaps should they, given the alternative options, such as reentering previously harvested stands after a short time or taking undersized trees. Given that many of these small scale timber operations must continue to function within the confines of their realities, I see the following points as potential recommendations for smallholder timber management following RIL guidelines.

Logging Block Size and Implications for Pre-harvest Inventories

A diversity of smallholder systems has been described in this review, illuminating the need to consider details of each one (such as logging block size) when planning timber harvests. In the state of Acre alone, several logging block sizes are used, from the four-ha harvest compartment in Pedro Peixoto, to the 10- and 50-ha systems of Porto Dias and São Luis de Remanso, respectively. As logging operations are currently executed, I believe that the 50-ha system is best suited to maintain timber species population viability simply because managers implement a pre-harvest inventory in a larger unit, providing a more accurate assessment of species distributions than with a 10-ha block (see Table 2-1). Also at this scale, logging crews will not need to reenter the 50-ha area as frequently as is required in the Porto Dias system, in which skidders often return annually to previously harvested logging blocks, reopening primary skid trails. Also, by harvesting

just one 50-ha unit per year, the logging crew is arguably exerting less effort in person hours and equipment usage, as compared to harvesting several non-contiguous 10-ha blocks in a given year.

Even so, political, cultural and socio-economic reasons exist for choosing one system over another. As such, although harvesting a 10-ha block in one year may not be the most ecologically viable option, it is what certain communities have to deal with, along with all of the associated environmental disadvantages. The 10-ha logging block system is also likely to grow in usage, as it is reportedly being favored by new communities wishing to manage for certified timber in Acre (N. Marcondes, *pers. comm.*). This reality suggests the need for full scale inventories of the entire landholding projected for timber management to minimize potential dangers of inbreeding and inadequate seed dispersal. The measurement of genetic variation is impractical for forest managers, but the potential for genetic drift can be reduced by basic knowledge of species populations generated by a landscape scale inventory (Jennings et al. 2001). Failing implementation of a 100% survey for those trees above at least the minimum diameter cut, rapid assessment inventories should be considered.

Botanical identification is a critical component of logging inventories, for which local parataxonomists are key partners. Before inventories take place, however, common names should be agreed upon, so that the same nomenclature is applied throughout the survey process. To ensure that these common names then correspond with their correct scientific counterparts, care must be taken to have samples identified with known herbarium examples, if possible. At the very least, care should be taken to not solely rely on databases generated by other organizations, whose own tree identifiers may also be

using common names that differ from those being used by a local parataxonomist from the timber project.

Monitoring

Recognizing that smallholder and community forests must establish their own management strategies while building on RIL guidelines, I suggest that systematic post-harvesting monitoring become integrated into logging operations, so that the benefits of recommended practices (such as pre-harvest liana cutting) may be tested at the appropriate scale. I recognize that smallholders in the rural tropics have limited financial and technical resources, and that monitoring may be unfeasible for them, unless they perceive that proposed activities add to their livelihood productivity. Accordingly, Ghazoul (2001) concedes that given associated high costs, an adaptive management approach, entailing collaboration between biologists and local forest managers, would allow for sustainable management to be based on the technical skills, ecological knowledge base, equipment, and funds available to the managers. This collaborative approach to forestry requires considerable communication between local managers and professional forestry technicians and an assessment of existing local ecological knowledge and traditional management systems (Carter 1996).

To this end, collaboration between smallholders, the agencies that support them and researchers who may be able to contribute technical expertise to the process should be fully encouraged. I recognize that working outside of traditional field sites in community settings obliges researchers and practitioners to approach forest management and monitoring differently, requiring them to consider social, political, and economic factors that affect the way that local peoples interact with the natural resource base. Beyond this consideration, we need to also think about the dynamics of the forest managed for a suite

of products and services at the scale discussed in this review, providing communities and smallholders with valuable feedback on their management systems and further providing a link between biodiversity conservation and SFM.

Moving beyond RIL

It is indisputable that RIL methods are a vast improvement over the much more destructive and haphazard conventional selective logging practices that are, unfortunately, still the norm (Schulze 2003). Yet, forest management under RIL guidelines alone could be at odds with the SFM model, which demands forest management for a suite of goods and services as well as the maintenance of ecological functions (Fredericksen and Putz 2003). Arguably, it is this broader suite of outputs that many communities target in their forest management schemes. Rural communities depend on wildlife, NTFPs, watersheds, and soil properties for their livelihoods, underscoring the importance of looking beyond sustained timber production as the most important forest management objective. Even if seed sources and dispersal mechanisms remain intact, thereby (in theory) ensuring the viability of future timber regeneration (Baraloto and Forget 2004), the long term logging impacts on these other livelihood and conservation elements must not be overlooked. It seems particularly appropriate then to move beyond an emphasis on RIL guidelines for smallholders engaged in timber management, while still incorporating its basic tenets into practical application.

It is highly probable that the so-called sustainability of intensive timber management may not be known until at least three cycles of harvest have been completed (Poore et al. 1989). This observation bolsters Simberloff's (1999) suggestion that any proposed silvicultural system designed to maintain biodiversity while producing timber should be treated as a hypothesis, due to limited empirical evidence supporting its

viability. Until then, though, we must continue to focus on forest management research that is not just rooted in tree reproduction biology (Guariguata and Pinard 1998), but also consider the application of such investigations at the smallholder and community levels, necessarily incorporating the local ecological knowledge that is often overlooked by academic researchers yet when it is acknowledged. In this way, we might be able to avoid the pitfalls of relying on RIL as proxy for SFM, especially in terms of the smallholder and community settings, which necessarily require considerations beyond the biological context.

Table 2-1. Contrasting tree population parameters from inventories at two different spatial scales commonly used by smallholders. Shown are the results of a 50 ha inventory of commercial species stems ≥ 35 cm DBH from PAE São Luis de Remanso, Acre, Brazil, along with the range of values (5 and 95%) obtained by 500 simulations of randomly-placed 10 ha plots within the larger area. Species were included only if they were represented by at least 10 trees in the plot, and their botanical identification had been verified. Bold cases indicate that the distribution of values from the 10 ha simulations exceeded the actual value at 50 ha, suggesting that management decisions would change depending on the scale at which inventories are conducted.

Species	**Common Name**	**Total Density (ind ha^{-1})**		**Harvestable Density[1] (ind ha^{-1})**	
		50 ha	**10 ha**	**50 ha**	**10 ha**
Aspidosperma vargasii	Amarelão	0.92	0.5 – 1.3	0.36	0.1 - 0.5
Myroxylon balsamum	Bálsamo	0.31	**0.2 – 0.6**	0.15	**0.1 – 0.4**
Tetragastris panamensis	Breu-vermelho	4.62	**2.6 – 8.6**	0.82	**0.2 – 2.2**
Cedrela odorata	Cedro-vermelho	0.87	**0.3 – 1.7**	0.67	**0.3 – 1.3**
Amburana cearensis	Cerejeira	0.56	**0.1 – 0.9**	0.41	**0.1 – 0.9**
Apuleia leiocarpa	Cumaru-cetim	0.97	0.7 – 1.7	0.79	0.5 – 1.3
Barnabydendron riedelii	Guaribeiro	0.82	**0.3 – 1.3**	0.67	**0.2 – 1.0**
Tabebuia serratifolia	Ipê-amarelo	0.23	**0.1 – 0.4**	0.08	**0 – 0.2**
Hymenaea parvifolia	Jutaí	1.21	**0.7 – 2.1**	0.64	**0.2 – 1.2**

[1] refers to minimum size harvested, which varies among species from 45 – 60 cm DBH.

Table 2-2. An example of spatial aggregation among neotropical tree species harvested by communities in Acre, Brazil. Shown are the minimum distance, if any, for each species at which individuals (stems ≥ 35 cm DBH) are significantly clustered (Ripley's L-statistic exceeding 95% confidence interval, based on 1000 bootstrapped values at 5 m distance intervals using Besag's edge correction conducted in Ripley software package; Marcon 2005). Data are from a 50 ha inventory of commercial species from PAE São Luis de Remanso, Acre (Centro dos Trabalhadores da Amazonia, *unpublished data*). Species were included only if they were represented by at least 10 trees in the plot, and if their botanical identification had been verified.

Species	**Common Name**	**Spatial Distribution**	
		Clumped?	**Distance**
Aspidosperma vargasii	Amarelão	No	
Myroxylon balsamum	Bálsamo	No	
Tetragastris panamensis	Breu-vermelho	Yes	6 m
Cedrela odorata	Cedro-vermelho	Yes	8 m
Amburana cearensis	Cerejeira	No	
Apuleia leiocarpa	Cumaru-cetim	No	
Barnabydendron riedelii	Guaribeiro	Yes	15 m
Tabebuia serratifolia	Ipê-amarelo	No	
Hymenaea parvifolia	Jutaí	No	

CHAPTER 3
EVALUATING LOGGING DAMAGE TO FUTURE CROP TREES IN ACRE, BRAZIL: IMPLICATIONS FOR MANAGEMENT OF A CERTIFIED COMMUNITY FOREST

Introduction

The guiding principle of sustainability dictates that a commercial forest should be managed in a way that limits as much undesirable damage as possible to the ecosystem. This is essential for not only ecological reasons, but economic ones as well (Müller 1998). For example, it is well known that unplanned (conventional) logging results in unnecessary damage to the residual forest and reduces the chances for future timber production (Pinard et al. 1995). Johnson and Cabarle (1993) estimate that anywhere from 26 to 75% of a given tropical stand of future crop trees (FCTs) are injured during conventional logging operations. Damage to residual stands is often correlated with logging intensity, which varies greatly, depending on spatial and temporal factors, access, equipment, terrain, climate, supplemental treatments, and market acceptance of lesser-known species (Metzger and Schultz 1984, Sist et al. 1998, Putz et al. 2000).

The most common harvesting method in the tropics is selective logging, whereby only scattered trees of sufficient economic value are extracted from the forest. In unplanned operations, selective logging has been observed to cause extensive canopy cover removal (Johns et al. 1996, Webb 1997, Whitman et al. 1997, Pereira et al. 2002), soil compaction (Whitman et al. 1997), long-term changes in tree species composition (Thiollay 1992), and reduced faunal diversity (Johns 1991, Thiollay 1992). Accordingly,

harvesting techniques and intensities need to be controlled to reduce residual stand damage (Webb 1997).

Controlled selective logging, or reduced-impact logging (RIL), utilizes methods that minimize damage to the residual stand, such as pre-harvest inventory, mapping, directional felling, vine cutting, and planning of skid trails, log decks, and roads (Uhl et al. 1997), potentially avoiding some of the post-logging damage noted by other authors (Pinard and Putz 1996, Johns et al. 1996). RIL techniques have been shown to reduce damage to residual stands in mixed dipterocarp forests in Southeast Asia by as much as 30-50% (Pinard and Putz 1996, Bertault and Sist 1997, Sist et al. 1998). Furthermore, RIL methods have demonstrated an increased efficiency of logging operations and a reduction in the amount of timber wasted during a typical harvest (Barreto et al. 1998, Holmes et al. 2002).

Even with implementation of RIL methods, sustainability of harvesting will ultimately depend upon advanced regeneration to replace the larger size classes during the next cutting cycle (Putz et al. 2000). This assumption is challenged by the damage incurred by the residual stand during logging operations. For example, Jackson et al. (2002) report that RIL operations in a FSC-certified timber concession in Bolivia killed or severely damaged an average of 22 trees (≥ 10 cm dbh) for every one extracted. In the same forest, Krueger (2004) found that flagging reduced damage to FCTs by 20% in felling gaps and by 10% along skid trails receiving 2-10 skidder passes, demonstrating that marking, or flagging, of FCTs can reduce damage to the residual stand by improving visibility during on-the-ground operations (Dykstra and Heinrich 1996). Currently, general Brazilian forest management practices do not employ this method.

Mechanisms underlying these residual stand impacts need to be understood in the context of specific local conditions (Sheil and Van Heist 2000). Commercial species distribution and human exploitation varies greatly between neotropical regions, supporting the argument that silvicultural treatments appropriate in one location will not necessarily be applicable in another (Putz 1996). For example, Webb (1997) points out that difficulties encountered in *Carapa nicaraguensis* forest swamp in Costa Rica required management techniques not employed in other logging operations. Given the need to develop and evaluate site-specific management strategies that will mitigate the negative effects of timber harvesting, it is essential to look at several variables across locales.

One forest type on which little has been written about in the management literature is the bamboo-dominated forest of southwestern Amazonia, which covers almost 180,000 km^2 (Griscom and Ashton 2003). This forest is characterized by a mix of structurally heterogeneous stands without bamboo and patches of trees scattered within a canopy dominated by dense stands of *Guadua* species (with some species of *Guadua* reaching heights of 30 m) (Griscom 2003). Similar to other mast seeding bamboo genera (Janzen 1976), *Guadua spp.* undergo a single synchronized reproduction event followed by synchronized mortality, at 25-30 year intervals (Nelson et al. 2001). At least two species, *G. sarcocarpa* and *G. weberbaueri* have been linked with reduced forest basal area as well as tree species richness (Silveira 2001, Griscom 2003). This phenomena is strongly linked to the species' ecology: branches of *G. sarcocarpa* and *G. weberbaueri* culms are endowed with curved barbs which act as grappling hooks (see Fig. 3-1), enabling the bamboo to be partially or fully supported by trees for vertical growth (Griscom 2003),

weighing them down, and increasing the likelihood of blow-downs (Griscom and Ashton 2003). In concert with the common tree blow-downs in this region, Griscom and Ashton (2003) propose that low recruitment of trees into the larger size classes explain the persistence of *Guadua* domination, rather than the catastrophic disturbances of fire and agricultural intervention cited by other authors. If *Guadua* achieves dominance in managed forests, commercial trees may become competitively excluded and a substantial proportion of the remaining stems may become deformed, due to mass loading (Griscom 2003). The low visibility in some areas of the *Guadua* dominated forest also presents an impediment to sawyers trained in directional felling (F. Correa da Cunha, *pers. comm.*). Combined, these factors present some unique challenges for silvicultural management and timber extraction in *Guadua*-dominated forest.

This study evaluated logging damage to FCTs in a certified community forest in Acre, Brazil. The main objectives of this investigation were (1) to determine the extent of area disturbed (canopy gaps and skid trails) during a small-scale certified logging operations, (2) to determine a difference in damage incidence to FCTs between forest with and without bamboo (*Guadua*), (3) to determine if likelihood of FCT damage is increased when lianas are present, (4) to determine to what extent harvesting can be conducted more intensely (m^3ha^{-1}) without incurring greater residual stand damage in general and damage to FCTs ≥ 20 cm dbh in particular, and (5) to determine to what extent marking of FCTs diminishes damage and/or mortality of FCTs (≥20 cm dbh) during logging.

Site Description

The Porto Dias Extractive Settlement Project (PAE) (S 10°00’39, 9”, W 66°46’26,4”) is situated in the northeastern corner of the Brazilian state of Acre, in the

municipality of Acrelândia. It is bordered by various rubber estates, cattle ranches, and colonization projects (see Fig. 3-2), and is separated from Bolivia by the Abunã River (Stone 2003). The site is defined by red-yellow latisols of low fertility and relatively flat topography, with an annual precipitation of 1890 mm yr^{-1}, most of which falls between November and March. As of 2001, approximately 97% of the reserve was still forested (CTA, *unpublished data*), but it is one of the last large contiguous pieces of forest in the municipality (N. Marcondes, *pers. comm.*). As such, it is under constant pressure from the influences of agricultural conversion within the PAE as well as illegal clearing on its borders. Older residents recall *Swietenia macrophylla* (mahogany) being logged in the region and transported via the Abunã River (Stone 2003), but mahogany is currently extremely rare in the PAE. Three tropical humid *terra firme* forest types are generally recognized in the reserve: open canopy forest; open canopy forest mixed with arborescent bamboo (*Guadua spp.*); and closed canopy forest.

Three species of *Guadua* are found in the study site: *G. weberbaueri*, *G. sarcocarpa*, and one as-yet unnamed species, which occurs in riverine forest along the Abunã River (M. Silveira, *pers. comm.*). At the end of 2004, when logging operations commenced, *G. sarcocarpa* underwent a monocarpic dieoff that affected much of the study area. Both *G. weberbaueri*, and *G. sarcocarpa* are thought to be synchronized on a 30 year masting cycle across large areas (100-100,000 km^2) (Nelson et al. 2001). The two species exhibit similar autecological characteristics, with an apparent lack of seed dormancy, germination occurring in both gaps and non-gaps, and partial shade tolerance among seedlings (Griscom 2003). In at least one study in Madre de Dios, Peru, *G. weberbaueri* demonstrated a lower mean stem diameter (cm) in comparison with *G.*

sarcocarpa (4.04 vs. 6.67, respectively), but exhibited higher stem density (stems/100 m^2) (34.2 vs. 23.8) (Griscom 2003).

The reserve encompasses approximately 22,145 ha, and is divided among *colocações*, or individual family landholdings constructed around pre-existing rubber trails (approximately 300 ha each). There are around 100 registered families who live in the PAE, in addition to an estimated 20 families who are squatting illegally in the reserve (Santos 2000). Traditional family livelihoods in PAE Porto Dias include forest extraction, hunting, subsistence agriculture, and small-scale animal husbandry (Stone 2003). Cash was customarily earned from rubber and Brazil nut harvests, but in recent years rubber tappers have been turning to logging as an alternative income source. The Center for Amazonian Workers (CTA), a non-governmental organization based in Acre, has been instrumental in developing the management framework of the Multiple-Use Community Forest Management Project. The project was initiated in 1995, but the first timber harvest was not harvested until 2000. SmartWood certified the logging operation using Forest Stewardship Council (FSC) criteria in 2002, distinguishing it as the second FSC-certified community timber management project in Brazil. FSC certification requires that the project implement a special monitoring program in the bamboo-dominated forest, as there is concern that logging damage may be greater there than in forest lacking bamboo (N. Marcondes, *pers. comm.*). Additionally, *Guadua*-dominated stands have been identified as important habitat for many valuable wildlife species, making them focal areas of interest for conservation efforts (Griscom 2003).

Forest Management and Harvesting Operations

Landholdings (approximately 300 ha each) of reserve residents serve as the timber management areas, and therefore do not favor the compartmentalization typical in

Brazilian industrial timber operations (Braz and Oliveira 1996). In compliance with federal regulation, 10% of the landholdings may be cleared for agricultural activities and 5% must be set aside for preservation (no harvesting of trees). Thus, most residents have at least 255 ha available for timber management, result in a 25-year logging rotation (255 ha/10 ha yr^{-1}= 25.5 years). The original ten members of the project all harvested 10 ha per year, but within a few years after the first timber harvest, it was decided that only five landholdings per year would be harvested. In the latter case, project members began harvesting every two years in their respective landholdings. However, given that some families have since dropped out of the timber project, other members have still harvested in sequential years (i.e. more than one or two years in a row), further confounding the question of the cycle length.

Inventories and mapping of all tree species ≥35 cm dbh are carried out one year prior to harvesting in 10 ha blocks within each landholding to be logged; as of 2005, though, a 100% inventory of the entire property had not been executed in any of the participating landholdings. After selecting the crop trees, the project must submit its logging plan to the Brazilian Institute of the Environment and Renewable Resources (IBAMA), which will then approve or disapprove the plan, based on an acceptable basal area removed, desired residual stand species distribution and crop tree proximity to water sources. Only trees ≥ 45 cm dbh may be legally harvested. Logging is generally carried out in the driest part of the year, from July to November. Twelve major timber species are presently exploited (see Table 3-1), with a low intensity of cutting (1-3 trees ha^{-1} or approximately 10 m^3 ha^{-1}). Since the timber project members began logging biannually, the total annual harvest for the project is around 500 m^3, or 100 m^3 $landholding^{-1}$. Sawn

board and artisan products from the community sawmill are primarily sold to the domestic certified market in Southern Brazil. The project does not market logs.

Members of the timber project (as of August 2005 there were seven participating families) conduct most of the logging activities. These techniques include inventories, selection of crop trees, assessment of defects in crop trees, directional felling, and skid trail planning, RIL skills for which they have been extensively trained over the last ten years by the Technological Foundation of Acre (FUNTAC) and the Tropical Forest Foundation (FFT) in Pará, and for which they now train other forest managers. The only activity that they do not directly participate in is skidding logs to the sawmill, a process contracted to heavy equipment owners and operators from outside the community. During logging operations, the landowner typically assists the construction of skid trails by flagging the path that the skidder or tractor operator will use.

Methods

Future Crop Tree (FCT) Selection

Fifty-two tree species were selected for the study, based on their potential importance as commercial timber sources (see Table 3-1; Ribeiro et al. 1999, Lorenzi 2000). Due to external market demands, however, only about 10-12 species are exploited on a regular basis in Porto Dias, most commonly *Amburana cearensis*, *Cedrela spp.*, *Peltogyne spp.*, *Dipteryx spp.*, *Apuleia leiocarpa*, *Tabebuia spp.*, *Aspidosperma vargasii*, and *Hymenaea intermedia*. For purposes of this study, the definition of FCTs was broadened to account for all species of commercial value, both current and future (see Table 3-1). When proper field identification was questionable, species verifications were conducted at the Federal University of Acre herbarium in Rio Branco, Acre, Brazil, but some trees were only identified to genus (see Table 3-1).

Experimental Design and Pre-logging Inventory

The greatest amount of damage to FCTs during logging occurs in areas closest to felled crop trees and skid trails (Johns et al. 1996). Therefore, observations of logging impacts were concentrated in a 50 m radius (hereafter referred to as "zone of impact") around each designated harvest tree in each of the 10 ha (200 x 500 m) logging blocks in four landholdings scheduled to be harvested in 2004-2005 (Barrinha 1, Barrinha 3, Palestina, and São Pedro). This number was eventually reduced to three, as the rainy season began before São Pedro could be logged. A full inventory of all commercial FCTs ≥20 cm dbh was conducted in the zones of impact. Sample sizes in the logging blocks (i.e. the number of zones of impact) depended on the abundance of crop trees, with a total of 18 zones of impact and 161 FCTs in Barrinha I, 20 zones of impact and 216 FCTs in Barrinha III, and 12 zones of impact and 109 FCTs in Palestina, including some overlap between zones. In 2004, before logging operations commenced, the following data were noted for FCTs: species, location (x, y coordinates), dbh (1.3 m or above the buttresses); total estimated height; qualitative trunk and crown quality ("good", "tolerable", or "inferior"); crown position ("dominant", "intermediate", "suppressed"); presence in bamboo-dominated stand ("yes"/"no"), and presence of at least one liana to the canopy ("yes"/"no").

Area Disturbed

All primary and secondary skid trails and treefall gaps were mapped using a compass and meter tape. The size of treefall gaps was estimated using the "center-point" system (modified from Runkle 1992). From a point in the approximate center of each treefall gap, compass angles in eight different directions were measured to the edge of intact canopy (Johns et al. 1996). This information was then entered into ArcGIS 9.0

(ESRI, Redlands, CA), to calculate area of the resulting polygon. The ground area transformed by the total length of skid trails was determined using modified methods from Johns et al. (1996), with skid trail width being measured at the beginning and end of each straight segment of trail. Trail width measurements were also taken at each bend to account for potential bulldozer maneuvering. This information was then combined with total skid trail length (estimated using pre-harvest inventory maps) in ArcGIS 9.0.

FCT Location in Bamboo (*Guadua*)-dominated Forest

Zones of impact (both in areas of high and low harvest intensity) naturally fell in locations where bamboo was present and in areas where it was absent. Location in *Guadua*-dominated stands was defined as at least one culm of *Guadua spp.* within 2 m of the FCT trunk.

Liana Presence

To test the effect of presence of lianas on FCT damage probability, liana occurrence on a FCT was denoted as at least one vine connecting to the canopy.

Harvest Intensity

To detect the impact that a range of timber harvest intensities could have on FCTs in Porto Dias, zones of impact were established in locations scheduled to receive a locally high intensity of cutting (≥3 trees ha^{-1}), and also in areas of locally low intensity cutting (1-2 trees ha^{-1}). This design allowed for establishing a harvest intensity gradient in each logging block, but did not necessarily allocate the treatments evenly across the logging blocks, as the abundance and distribution of crop trees varied greatly between landholdings.

Marking Treatment

To test the effect of marking, about half of all inventoried FCTs in each of the four logging blocks were marked. All FCTs within the zone of impact of a tree to be harvested were either marked or left unmarked, stratifying the treatment between regions of low and high crop tree density (see Figs. 3-3-5). Marking entailed a single band of orange paint 20 cm wide at approximately 1.7 m above the ground (adapted from Krueger 2004).

Initially, the random assignment of the marking treatment was considered when designing this experiment. However, it was decided that the highly visible presence of even one marked FCT in the vicinity of unmarked FCTs could induce the logging crew to alter their activities in some way as to favor neighboring FCTs. This potential source of bias suggests that statistical independence of the FCT treatments would have been difficult to achieve even with this type of experimental design.

Assessment of FCT Damage

Inventory data were used to construct a spatial database in ArcGIS 9.0. For each FCT, distance to nearest felled tree, nearest gap edge and nearest skid trail were measured. Harvest intensity (trees ha^{-1}) was estimated for each FCT as the volume exploited in the surrounding 1 ha (circumscribed by a radius of 56.4 m). Local FCT density was estimated as all FCTs $\geq$ 20 cm dbh within a 25 m radius of a given FCT. This measurement was included to control for variation explained by local density when testing for other factors such as marking. Seven months after logging activities were completed, in June and July, 2005, all previously inventoried FCTs were revisited and classified according to levels of damage or mortality (see Table 3-2).

Statistical Analyses

Quantifying FCT damage was considered for this study. A logistic regression analysis was used to test the binary response variable (damaged/not damaged) by treating each FCT as an independent statistical unit to which a series of independent variables was assigned: local harvest intensity (continuous); distance to nearest felled crop tree (continuous); distance to nearest skid trail, (continuous); distance to nearest gap (continuous); neighborhood FCT density (continuous); presence of bamboo (binary); marking (binary); and presence of lianas (binary). A regression analysis was performed to determine the influence of crop tree dbh on felling gap area. Differences were considered significant at $P \geq 0.05$.

Results

Overview of the Logging Operation

The mean tree (≥ 35 cm dbh) basal area was determined to have the following values for the three landholdings: 7.7 m^2 ha^{-1} in Barrinha I, 8.3 m^2 ha^{-1} in Barrinha III, and 8.8 m^2 ha^{-1} in Palestina. All harvesting occurred between October and November 2004, and a bulldozer (Valmet 128) was used to extract logs. Two community-based sawyers participated in the operations: one felled trees only in Palestina and the other cut in both Barrinha I and Barrinha III. No new roads were created to facilitate logging operations, and pre-existing skid trails in logging blocks from the project's previous harvesting activities were reopened to access new areas. All three logging blocks were located close to existing roads, so boles were eventually skidded to log decks adjacent to these routes.

A total of 22.3 ha of 30 ha were sampled in the three logging blocks. Forty five trees were extracted from all of the landholdings, with 38 actually cut and 7 that fell

under natural circumstances at least one year prior to harvesting, for a total harvested volume of 237.7 m^3, or approximately 7.9 m^3 ha^{-1}. Felled tree (n=38) dbh (cm) ranged from 60-114, with a mean of 71.7 (s.d. 14.73). Average harvested tree density was approximately equal between the three logging blocks (see Table 3-3), suggesting that any differences in damage incurred by FCTs was not due to differences in logging intensity (Johns et al. 1996). Some of the trees originally selected were not cut for various reasons (usually they were found to be hollow prior to harvesting), so loggers harvested several substitute trees. As a result, some of the trees designated as experimental FCTs the previous year were not affected by logging operations, as they were far from any skid trails or felling gaps (≥ 60 m from felling gap, cut stump, or skid trail). These trees (a total of 18) were not considered for the analysis. At the same time, damaged trees that were near the newly substituted crop tree sites were not tallied after harvesting, because no pre-logging data were gathered for them.

FCT Damage

A total of 70 FCTs out of 468 were damaged during logging operations, or approximately 15% of the experimental population. The most common damage types were slightly damaged crowns, followed by scraped boles and roots, classified as "minor" damage (see Table 3-3). Tree felling operations damaged a greater proportion of trees than skid trail construction and use, but the results were not statistically significant.

Marking did not significantly reduce overall FCT damage in the Porto Dias operations (see Table 3-4). Furthermore, the incidence of FCT damage levels was not higher in *Guadua*-dominated forest, nor was there a significant interaction between a FCT's presence in a *Guadua*-dominated stand and the marking treatment (interaction term in Table 3-4). Fifty-six percent of the inventoried FCTs carried lianas, but damage

probability due to liana presence on the FCT was not highly significant (P = 0.10), but many damaged FCTs had at least one large liana attached to its crown (67%).

The closer a FCT was to a skid trail or logging gap, the greater the likelihood of it being damaged (see Table 3-4). In contrast, distance to cut stump and local FCT neighborhood density had no significant effect. Harvest intensity (see Table 3-4) also did not influence the probability of FCT damage, implying that in this study, the probability of FCT damage did not increase with the number of trees harvested in the surrounding hectare.

Area Disturbed

The percentage of total sampled area affected by harvesting was 28.9%, with the majority in felling gaps (16.8%). Average area per felling gap amongst all three logging blocks (mean ± s.d.) was estimated to be 393 m^2 ± 181. Regression analysis revealed a significant positive effect of crop tree dbh in single tree gaps area (see Fig. 3-6). Skidding damaged 11.8% of the total surface area.

Discussion

Timber harvesting, as conducted by the Multiple-Use Community Forest Management Project in PAE Porto Dias, had minimal impacts on residual stands, at least on commercial individuals ≥ 20 cm dbh. Additionally, potential damage was avoided by not having to construct new log landings and log roads. Even so, the high values reported for area damaged, both skid trails and felling gaps, merit analysis (see Table 3-3). The values for felling gap area per tree cut were in some cases almost twice that of other reported neotropical studies (see Johns et al. 1996). In many instances, increased gap sizes can be explained by the size of the trees that were felled. For example, one of the most valuable taxa in Porto Dias, *Dipteryx spp.*, comprised some of the largest

individuals amongst harvested trees (up to 114 cm dbh), contributing to the strong positive correlation between logging gap size (m^2) and dbh of crop trees (see Figure 6). In the case of Palestina, more than 60% of the harvested trees were ≥ 70 cm dbh, explaining in part why this particular logging block had the greatest mean gap size (433 m^2) amongst the three landholdings. When considering exceptionally large trees for harvest, forest managers might do well to consider the tradeoffs between creating a very large gap (thereby potentially increasing fire and windthrow risk) and the substantial profit for the sake of the wood.

It is also worth noting that only those species valued for their merchantable timber were inventoried for the experiment. For example, locally important NTFP species such as *Bertholletia excelsa* and *Hevea brasiliensis* were not tallied during the original inventories, but during the post-logging survey, damage to several trees of these species was observed. For future studies, it would be prudent to address some of the issues of impact of timber extraction on other livelihood activities, such as NTFP collection, particularly in such regions as the Brazilian Amazon where many communities are attempting to integrate a suite of income-generating activities. Commercial logging, with potentially high financial yields, is unlikely to disappear in community forests. Nevertheless, interest persists in the harvesting of non-timber products such as Brazil nuts, rubber, and *Copaifera* and *Carapa* oils, species that may be adversely affected by timber harvests. When considering these factors, can timber harvesting and other livelihood strategies be made more compatible and if so, what are the tradeoffs (see Salick et al. 1995, Romero 1999, Menton 2003)?

FCT Damage in *Guadua*-dominated Forest

Given the low visibility in *Guadua*-dominated forest, I expected that FCT damage rates would be higher in these areas, but results proved otherwise. There could be a number of explanations for this outcome: even though poor visibility in *Guadua* groves has been cited as an impediment in the felling of large trees because neighboring FCTs are not always easily seen, FCT damage might be minimal simply because there are fewer FCTs to damage, as demonstrated by the low basal area estimates cited by other authors (Silveira 2001), or, more specifically, by the low representation of smaller stems in those same estimates. *Guadua* frequently outcompetes tree seedlings and saplings such that they fail to recruit into larger size classes (Griscom 2003). Therefore, one could conclude that there are comparatively more FCTs from the lower diameter classes to damage in forests without bamboo, or at the very least, there are more trees that operators must avoid when skidding or felling. It is also uncertain as to how the synchronized mortality of *G. sarcocarpa* may have directly or indirectly affected felling operations, although visibility may have improved in these stands because mature bamboo culms were drying out and falling to the ground in some areas.

Even though I did not find that selective logging increased the probability of FCT damage in *Guadua*-dominated stands, the potential for other types of long-term residual stand damage warrants future investigations. We observed dramatically increased light levels in post-logging bamboo-dominated forest, possibly attributable to the loss of the biggest canopy trees in the block, of which there were few to begin with. There are a few potential problems associated with this. First, by opening large areas during harvesting, the chances of repeated small-scale disturbances, such as windthrow and fire, are increased (Webb 1997). Given that *Guadua* benefits from intermediate and large canopy

disturbances (Griscom 2003), once a gap is opened, bamboo may proliferate and competitively exclude the majority of tree stems. One example of this phenomenon is Whitmore's (1984) observation of the reduction in value of Malayan forests that were invaded by arborescent bamboo after unplanned logging operations. The high light intensity characteristic of large gaps might also select for a suite of different species as compared to small gaps (Denslow 1980), a dilemma if the regeneration of small gap-preferring species is desired by the forest managers. In general, *Guadua*-dominated stands are already characterized by a discontinuous canopy, which may amplify the effects of openings created by logging activities (i.e. increased light levels). As such, for future operations, forest managers may want to select crop trees based on the presence of large neighboring trees, ensuring that some sort of basic canopy structure is kept intact around the logging gap if smaller openings are desired.

Another concern in this forest type is the potential for high grading, or the type of harvesting that selects for the largest and most valuable individuals. One of the underlying assumptions of RIL is that advanced regeneration of valuable commercial timber species exists uniformly throughout the stand so that future harvests are not compromised by the present one (Schulze 2003). Yet in *Guadua*-dominated stands, the lack of advanced regeneration potentially increases the chances of having small, poorly-formed trees of lesser commercial value dominate the next harvest. For the operation described in this paper, FSC certification standards, and indeed, Brazilian federal law, require that forest managers leave at least 10% of the harvested species' populations as seed trees. This perhaps ensures harvests for generations to come, but it does not guarantee a sustainable timber yield for the immediate second cycle. As well, even

though Brazilian federal law imposes a minimum diameter cut of 45 cm dbh, many species in this operation are not harvested until they are much larger (see Chapter 2), since taxa such as *Dipteryx spp.* are characterized by high levels of sapwood and thus attain commercial value at a greater girth. As a result, there is often an understandable temptation to take the biggest and the best individuals from a site, although at least one landowner in Porto Dias admitted hesitation in taking large *Dipteryx spp.* individuals, as they are often the most prolific seed sources (F. Correa da Cunha, *pers. comm.*). The complexities associated with timber harvesting in *Guadua*-dominated forest may require a much different approach to silvicultural techniques than developed in other parts of Amazonia, and underscore the need for future monitoring of harvesting activities and residual stand dynamics as well as extreme caution in the removal of the largest trees on site.

Impact of Lianas on FCT Damage

Lianas connecting the crowns of trees will often cause a domino-like effect during felling operations (Vidal et al. 1997), although few adjacent trees were completely brought down by felled trees during this operation. Other neotropical studies (see Fox 1968, Appanah and Putz 1984, Johns et al. 1996, Vidal et al. 1997, Pereira Jr. et al. 2002, Gerwing and Uhl 2003), have demonstrated increased damage during logging operations when vines were present. Although I expected vines to increase FCT damage, I did not observe any statistically significant effects at the α=0.05 level, though given results in other studies, it is probable that the magnitude of the outcome is consequential from a management standpoint (Steidl and Thomas 2001). It was evident from post-logging observations in the field that parts of crowns were pulled down by bulldozers at they

moved through the forest. Undoubtedly, if lianas associated with FCTs had been cut at least one year prior to harvesting, some of this damage could have been avoided. Still, Porto Dias forest managers expressed reluctance to cut lianas prior to the felling cycle (*pers. obs.*), citing their importance as a food source for wildlife species. Yet, it has been shown that lianas negatively influence the growth, leaf production and sexual reproduction of host trees (Parren 2003), and when not cut prior to harvesting operations, increase the potential danger for logging crews (Amaral et al. 1998). Parren and Bongers (2001) propose that the benefits of cutting or not cutting lianas prior to logging is site dependent, so it may be that local forest managers must experiment with this particular RIL guideline before making a final decision as to pre-harvest silvicultural prescriptions.

Impact of Harvest Intensity on FCT Damage

Local harvest intensities in the logging operation described here removed 1-4 trees ha^{-1}, seemingly with no increased levels of residual stand damage in the ≥ 20 cm dbh class as the maximum ceiling of harvest intensity (4 trees ha^{-1}) was reached. This type of low-intensity selective logging is attractive for conservationists espousing the implementation of sustainable forest management as a way to limit canopy cover loss and protect ecosystem function and biodiversity (Whitman et al. 1997). Even so, Gullison and Hardner (1993) have pointed out that as the number of trees removed increases, fewer new skid trails are required (although the damaged area per tree due to logging gaps remains constant). In the case of the operation described here, even though there was no evidence supporting increased levels of FCT damage as harvest intensity increased, the average area (m^2) damaged per tree extracted due to skid trail construction was quite high (see Table 3-3) as compared to other neotropical studies, relative to the number of trees extracted per hectare. For instance, Johns et al. (1996) found that unplanned logging

operations using a bulldozer to skid trees damaged 119 m^2 per bole removed in relation to skid trails (with an average harvest intensity of 5.6 trees ha^{-1}). The high levels of area damaged per tree extracted in this study suggest that there might be a tradeoff associated with trying to harvest isolated trees across even a relatively small area of 10 ha, both in terms of cost and as a function of area and trees damaged.

One way to offset this effect of low intensity harvests would be to increase localized harvest intensity at a fixed volume. The primary benefits of this method would be reduction in mortality of smaller size classes as well as fewer skid trails, not necessarily reduction in mortality or damage to FCTs (Panfil and Gullison 1998). Another consideration is that large gaps allow for more solar radiation to reach the forest floor, drying remnant organic matter such as leaves and increasing the amounts of fuel in post-logging sites (Johns et al. 1996). Furthermore, the release thresholds of pioneer species, lianas and bamboo in the large gaps resulting from this scenario are unknown (although conditions might be ideal for the regeneration of such species as *S. macrophylla*) (Panfil and Gullison 1998). As such, the implementation of an increased localized harvest intensity is risky, especially for a landowner limited to a management unit of 300 ha.

Impact of Marking Treatment on FCT Damage

Despite positive results of marking FCTs in at least one other study (see Krueger 2004), marking did not appear to reduce FCT damage in the Porto Dias timber operations. That FCT marking had no effect on damage levels along skid trails was surprising, given that machine operators were from outside the community and therefore not as familiar with the forest. Even so, the owners of the landholdings were responsible for flagging the path of the skid trails prior to extraction, so as to avoid any damage

associated with skidding. This precaution could explain why marking had no effect on FCT damage along skid trails in this particular study, as bulldozer operators were already able to move without hesitation, especially when creating secondary and tertiary trails. As well, most logs were bucked to about 5 m prior to skidding, perhaps avoiding some of the damage that occurs when tree-length logs are skidded. Two local sawyers were also in charge of felling operations, perhaps adding to the effectiveness of the operation, as they are both experienced harvesters in Porto Dias, in addition to having lived for many years in the reserve.

Conclusion

As more traditional forest dwellers begin to incorporate formalized timber management into their livelihood strategies (which in the recent past have been based on the exploitation of NTFPs), it will become increasingly important to monitor and document these systems. Historically, most of the neotropical research and forest legislation has focused on large-scale operations (Oliveira 2000). Smallholders will necessarily differ from their industrial counterparts, given their low harvest intensities and management regimes that favor a suite of forest products and services beyond timber.

Even though many general sustainable logging guidelines exist, one of the greatest barriers to good silvicultural management is developing site-specific management objectives and monitoring principles that recognize the heterogeneity inherent in tropical forest systems (Putz 1996). As was shown in this study, the gap mosaic of the forest will change dramatically in comparison to its pre-logging state, even when low intensity, RIL guidelines are followed (Webb 1997). Accordingly, community forest managers must establish their own management paradigm while building on the RIL model. Given the series of poorly understood parameters outlined in this study (pre-harvest liana cutting,

post-logging stand dynamics in bamboo-dominated forest, etc.), it is suggested that systematic post-harvesting monitoring accompany logging operations (already one of the requirements for FSC certification), so that the benefits of recommended practices may be evaluated by the managers at the appropriate scale. Pretty and Hine (1999) argue that monitoring methods must be sensitive to local needs in order to work in the long term, primarily due to limited financial and technical resources. It stands to reason then that for monitoring purposes, local peoples' efforts may be better placed in protecting the larger size classes of FCTs, particularly if the concessions are on a relatively short (30 years in this case) cutting cycle. This particular study focuses on the adult and sub-adult size classes (≥ 20 cm dbh) in the hopes of capturing information about the next harvest generation. Many previous logging experiments have concentrated on seedling regeneration, but these seedlings are unlikely to be recruited to harvestable size classes within at least two subsequent harvest cycles (Clark and Clark 1992, Connell and Green 2000). There may be more of a solid commitment to participatory monitoring if the individuals in question could easily project a young tree's value across thirty years (which would be in their children's lifetime), as opposed to a seedling that may take up to 100 years to get to a harvestable size, if it survives at all.

Reducing damage is one of the most important components of a sustainable forest management program (Johns et al. 1996). Direct damage to FCTs was shown to be slight for this study, and indeed, it was considerably less than in other neotropical investigations. Even so, damage could be reduced even further. The Porto Dias timber management project is harvesting lightly (7.92 m^3 ha^{-1}), but the implementation of liana cutting, cautious selection of both crop and seed trees and avoidance of stands heavily dominated

by *Guadua* could all help approach sustainable forest management, a concept which may have the best chances of being executed by those who directly benefit from the natural resource base on a daily basis.

Table 3-1. Commercial timber species inventoried during 2004 logging operations in the PAE Porto Dias, Acre, Brazil. For those species harvested in 2004-2005, total volume (m^3 ha^{-1}) is given.

Scientific Name	Common Name	Family	Total volume harvested (m^3) 2004-2005
Amburana cearensis.	cerejeira	Fabaceae	56.6
Andira sp.	angelim branca	Fabaceae	
Apuleia leiocarpa	cumaru cetim	Caesalpiniaceae	23.4
Aspidosperma megalocarpon	carapanaúba preta	Apocynaceae	
Aspidosperma sp.	carapanaúba amarela	Apocynaceae	
Aspidosperma spp.	pereiro	Apocynaceae	
Aspidosperma vargasii	amarelão	Apocynaceae	9.7
Astronium lecointeii	aroeira	Anacardiaceae	1.8
Astronium sp. 1	aroeira preta	Anacardiaceae	
Brosimium uleanum	manitê	Moracae	9.4
Carapa guianensis	andiroba	Meliaceae	
Cariniana sp. 1	tauari churu	Lecythidaceae	
Cedrela cf .fissilis	cedro branco	Meliaceae	1.7
Cedrela odorata.	cedro rosa	Meliaceae	4.4
Clarisia racemosa	guariúba amarela	Moraceae	
Copaifera cf. langsdorfii	copaiba preta	Caesalpiniaceae	
Copaifera multijuga	copaiba branca	Caesalpiniaceae	
Couratari spp.	tauari	Lecythidaceae	
Diplotropis spp.	sucupira preta	Fabaceae	
Dipteryx sp. 1	cumaruzinho	Fabaceae	6.1
Dipteryx spp.	cumaru ferro	Fabaceae	65.0
Enterolobium shomburgkii	fava orelhinha	Mimosaceae	7.7
Geissospermum sp.	quariquara amarelo	Apocynanceae	
Hymenaea intermedia	jatobá	Fabaceae	
Hymenaea oblongifolia	jutaí	Fabaceae	19.5
Hymenaea sp.	jutaí da folha grande	Fabaceae	
Hymenolobium cf. excelsum	angelim preto	Fabaceae	14.1*
Hymenolobium sp.	angelim da mata	Fabaceae	
Hymenolobium sp.	angelim pedra	Fabaceae	
Hymenolobium sp.	favela preta	Fabaceae	
Jacaranda copaia	marupá	Bignoniaceae	
Manilkara spp.	maçaranduba	Sapotaceae	6.5

Continued Table 3-1.

Scientific Name	Common Name	Family	Total volume harvested (m^3) 2004-2005
Martiodendrom elatum	pororoca	Caesalpiniaceae	
Mezilaurus itauba	itaúba	Lauraceae	
Minquartia guianensis?	quariquara branca	Olacaceae	
Myroxylon balsamum	bálsamo	Fabaceae	2.5
Parkia pendula	angico vermelho	Mimosaceae	
Peltogyne paniculatum	roxinho da folha grande	Caesalpiniaceae	
Peltogyne sp.	roxinho	Caesalpiniaceae	
Pouteria cf. reticulata	abiurana preta	Sapotaceae	
Pouteria sp. 1	abiurana roxo	Sapotaceae	
Pouteria sp. 2	abiurana abiu	Sapotaceae	
Qualea cf. tesmanni	catuaba	Vochysiaceae	3.5
Simarouba amara	marupá preto	Simaroubaceae	
Tabebuia cf. impetigenosa	ipê roxo	Bignoniaceae	
Tabebuia spp.	ipê amarelo	Bignoniaceae	2.91
Tetragastris altissima	breu vermelho	Burseraceae	
Vatairea spp.	sucupira amarela	Fabaceae	
Vochysia sp. 1	cedrinho	Vochysiaceae	
Vochysia spp.	guaruba	Vochysiaceae	

* volume recorded under one common name, but likely reflects more than one species of *Hymenolobium*.

Table 3-2. Classification of damage sustained by FCTs (future crop trees) during logging operations (modified from Jackson et al. 2002 and Krueger 2004).

Damage type	**Bole**	**Root**	**Crown**
Severe	Snapped at base, bent or leaning	Uprooted	Loss ≥ 66% crown
Moderate	Exposed cambium	Exposed cambium tissue	Loss 33-66% crown
Minor	bark scrape	root scrape	Loss ≤ 33% of crown

Table 3-3. Summary of harvesting impacts in 2004 for the three logging blocks at PAE Porto Dias, Acre, Brazil.

Impact variable	Barrinha I	Barrinha III	Palestina
Area sampled (ha)	8.0	8.4	5.4
Harvest density			
Number of harvested trees ha^{-1}	1.5	1.6	1.4
Percent of harvested trees ≥ 70 cm dbh	27	50	64
Volume harvested (m^3 ha^{-1})	7.6	8.5	7.7
Disturbance dimensions			
Number of logging gaps	14	13	11
Felling gap area (excluding bole) (m^2 ha^{-1})	545.6	433.4	476.6
Proportion of area damaged due to felling gap (excluding bole) (%)	5	4	5
Mean gap size ± SD, excluding bole (m^2)	389.7 ± 119	361.2 ± 249	433.3 ± 167
Skid trail area (m^2 ha^{-1})	519.4	318.6	347.0
Proportion of area damaged due to skid trail (%)	5	3	4
Total area damaged (ha)	1.15	0.83	0.89
FCT damage rate			
Total number of damaged FCTs	18	37	15
Basal area of damaged FCTs (m^2)	0.34	0.63	0.48
Proportion of FCTs damaged (%)	12	17	15
Frequency of damage categories (%)			
Minor or moderate crown, bole, or root damage	78	70	80
Severe crown damage	6	11	0
Severe bole damage	22	24	7
Severe root damage	6	3	27

Table 3-4. Results of a logistic regression for the probability of damage to future crop trees (FCTs) at Porto Dias, Acre, Brazil during 2004 logging operations. Shown are the estimates for five continuous covariates, including distance to nearest stump, felling gap, and skid trail, neighborhood FCT density (number of stems within a 25 m radius) and local felling intensity (number of trees felled in the surrounding hectare); and for three categorical independent variables, including liana presence, location in bamboo-dominated forest, and marking treatment. Statistically significant variables at $P < 0.05$ are shown in bold.

Variable	**Estimate**	**Wald**	**p**
Log distance nearest stump	-0.07	0.01	0.91
Log distance nearest felling gap	**-1.48**	**22.96**	**≤0.001**
Log distance nearest skid trail	**-2.25**	**28.37**	**≤0.001**
Log neighborhood FCT density	-0.26	0.24	0.62
Local felling intensity	0.16	0.88	0.35
Liana presence (L)	0.31	2.79	0.09
Location in bamboo (G)	-0.02	0.01	0.90
Marking treatment (M)	-0.14	0.57	0.45
L*G	0.14	0.52	0.47
L*M	0.24	1.55	0.21
G*M	-0.10	0.29	0.59
L*G*M	0.11	0.32	0.57

Figure 3-1. Photograph taken in PAE Porto Dias, Acre, Brazil, demonstrating the re-curved thorn on a branch of a *Guadua* culm.

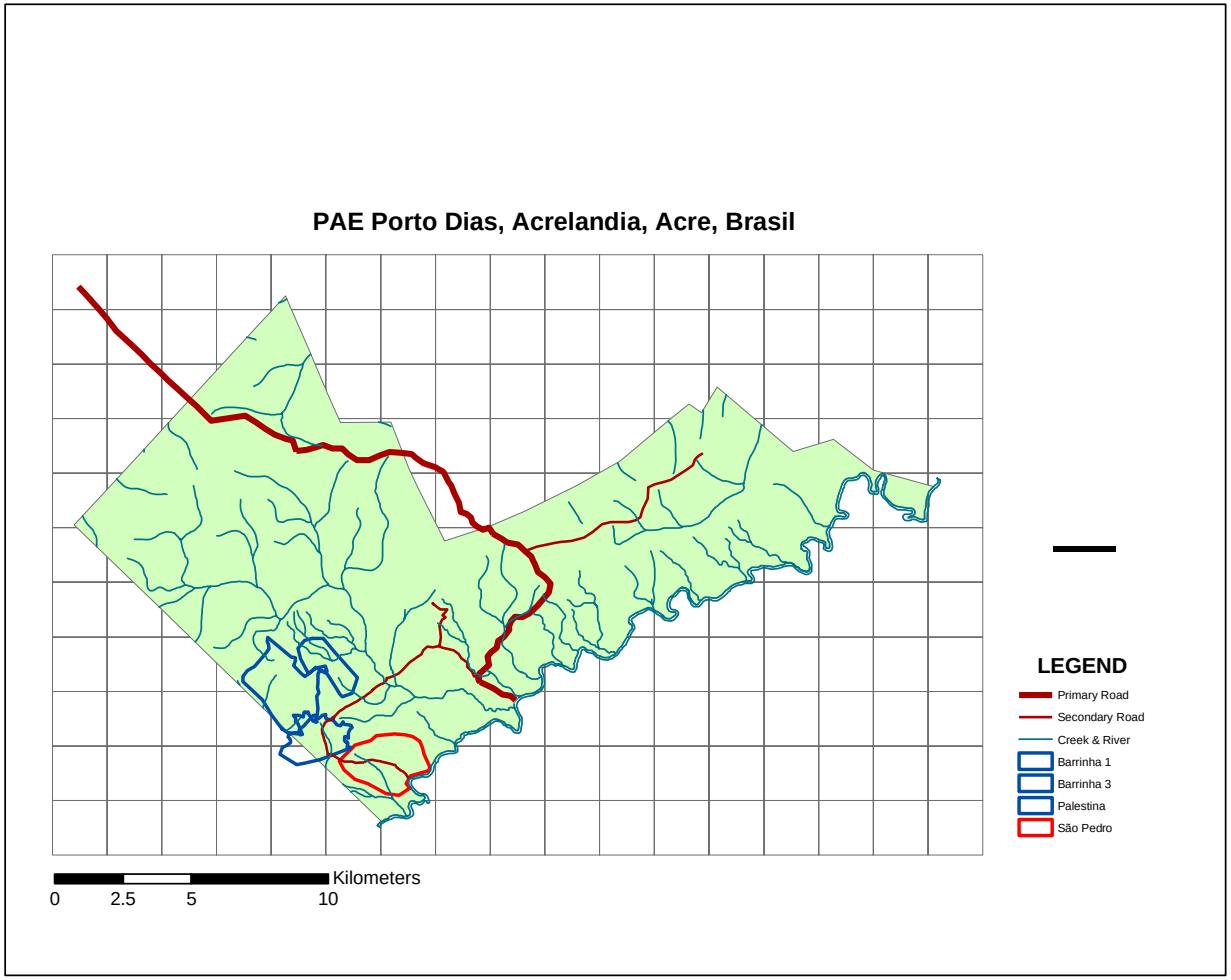

Figure 3-2. PAE Porto Dias, with the four landholdings inventoried in the study, including the three (in blue) harvested in 2004.

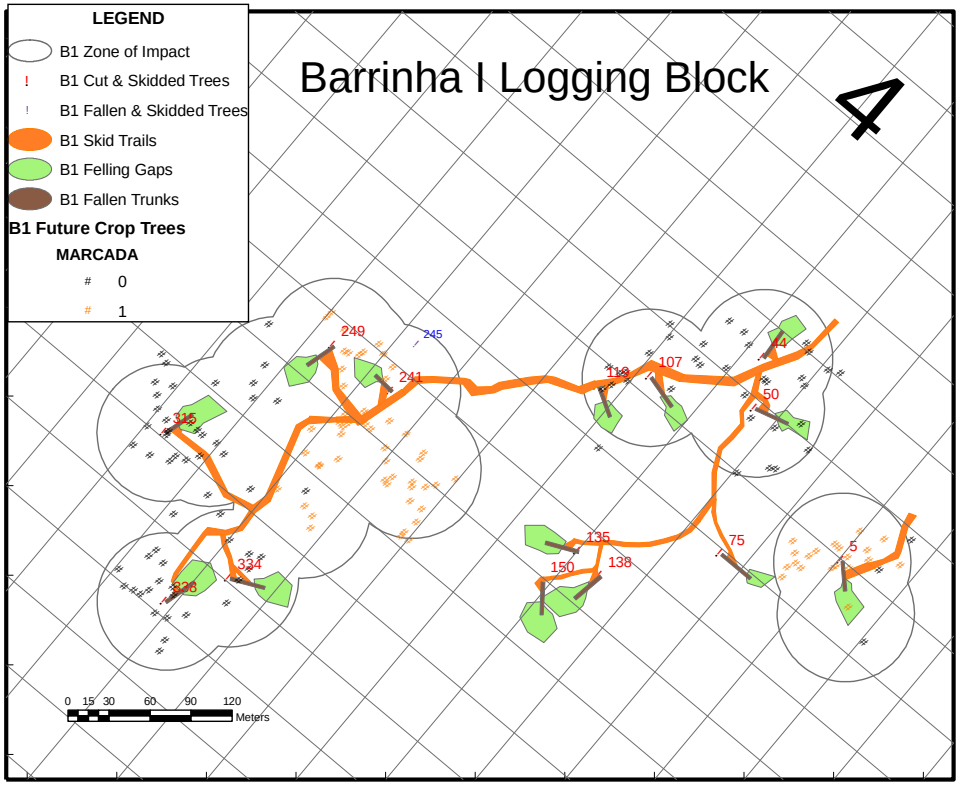

Figure 3-3. Logging block of Barrinha 1, showing skid trails, harvested trees, and location of future crop trees (FCTs) in the zones of impact.

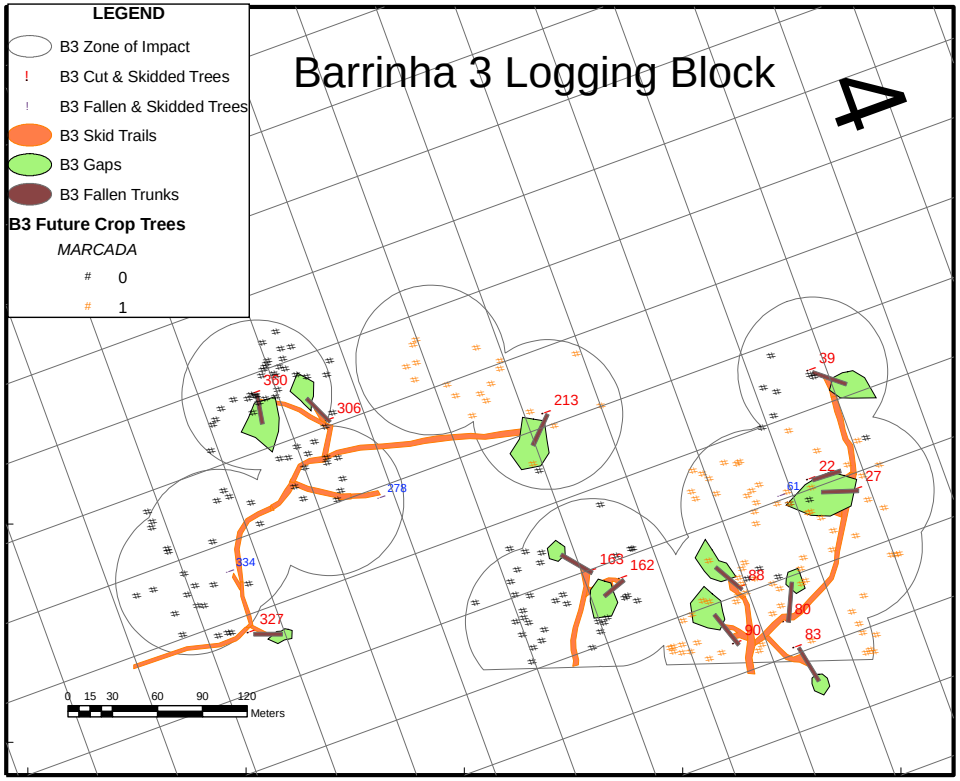

Figure 3-4. Logging block of Barrinha 3, showing skid trails, harvested trees, as well as location of future crop trees (FCTs) in the zones of impact.

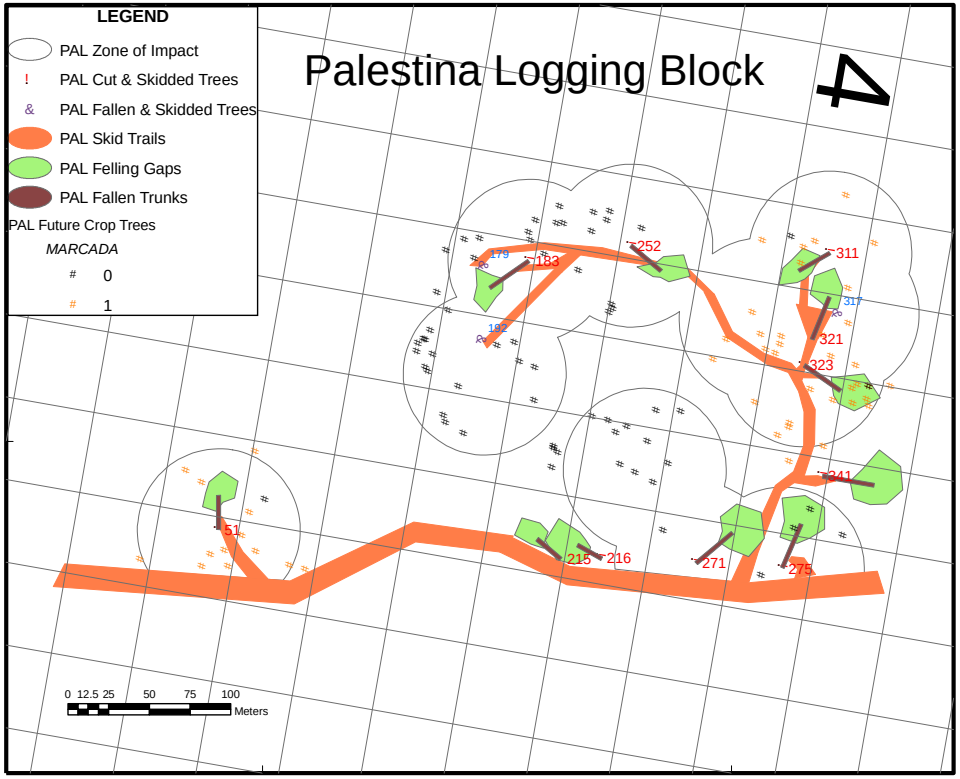

Figure 3-5. Logging block of Palestina, showing skid trails, harvested trees, as well as location of future crop trees (FCTs) in the zones of impact.

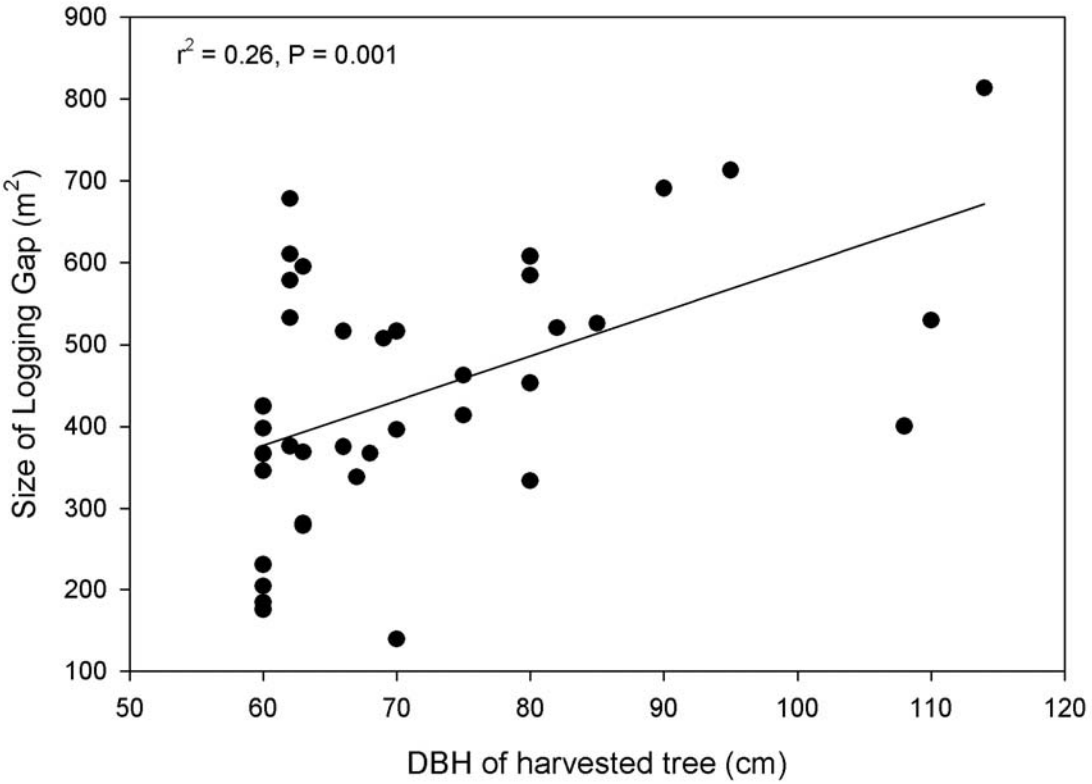

Figure 3-6. Relationship between size of felled trees and the surface area of the corresponding logging gap created during felling.

CHAPTER 4
CONCLUSION

Redford and Stearman (1993) propose that despite claims that low-impact extraction is compatible with biodiversity conservation, if a full range of genetic, species and ecosystem diversity is to be maintained, then any significant activity by humans must not be allowed. Advocates of SFM have countered this argument by stating that large-scale preserves have a limited chance of succeeding in the face of restricted financial resources, human population growth, and economic pressures (Putz et al. 2000, Pearce et al. 2003). Who is correct? Or is there no one right answer? Instead, should areas managed for timber resources be but one component of an interconnected model of complementary nature, production and indigenous reserves, allowing for various degrees of protection (Peres and Terborgh 1995)? The forest is going to be exploited, whether so-called preservationists like it or not. The extent to which that exploitation may incur permanent damage can be controlled in some ways by planned management activities. Even though a strong link between biodiversity conservation and SFM may never be fully satisfied, the global demand for tropical hardwood is not likely to subside anytime soon. This is especially true of Brazil, which is simultaneously the world's largest producer and consumer of tropical hardwood (May 2002).

Questions about economic and environmental sustainability still beleaguer community-based timber projects' quest for staying power, though many see the timber projects and forest managers as key assets in protecting forest ecosystems from encroaching mechanized agriculture and cattle production, two of the greatest threats to

Amazon biodiversity (Nepstad et al. 2002, Kainer et al. 2003). With future studies, it will be challenging to detect changes in the resource base given that many plant and animal species are harvested simultaneously in tropical forests; ecological tradeoffs associated with these multispecies systems are so complex as to warrant long-term trial-and-error experiments (Becker and Ostrom 1995). Thus, complex community level changes may be difficult to asses (Hall and Bawa 1993). Also, a critical factor to consider in this debate is such ambiguous vocabulary as "sustainable," "biodiversity" and "ecosystem management" (Ghazoul 2001). The inability of biologists to agree upon this lexicon makes the practical application of sustainability guidelines by community forest managers even more challenging. Consequently, there is a greater risk in resource degradation if specific ecological problems are not readily identifiable.

The conclusions reached in this thesis suggest that forest dwellers may be able to sustainably use common resources of the forest, but whether or not they do so will depend on a wider setting, that of a shifting national and international politico-economic climate (Cardoso 2002). Fortunately, particularly in the state of Acre, the government has been extremely supportive of forest dweller communities, in addition to providing a strong facilitative role. With the continued presence of external agencies, these communities will stand a better chance at conserving forest resources.

One issue that is frequently mentioned in the literature but still lacks substantial investigation is the idea that economic sustainability is not always consistent with ecological sustainability (Hall and Bawa 1993). The links between sustainable development and resource management must be studied further, as the rural poor do indeed have an invested interest in sustaining the resource base, but at the same time they

must also increase and diversify their overall livelihood (Cardoso 2002). Ultimately, the success of SFM will be contingent upon addressing the drivers of forest degradation and deforestation. Conservation will not occur through SFM and forest certification alone; local, regional, national, and international policy makers must also be engaged in this process. Otherwise, SFM as a tool for biodiversity conservation has little chance of success.

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BIOGRAPHICAL SKETCH

Cara A. Rockwell was born on May 25, 1971, in Belleville, Illinois, and grew up in Tallahassee, Florida. She received a Bachelor of Science degree in biological sciences from Florida State University in 1994. Following graduation, she traveled for a year in Asia, Australia and the South Pacific, which fueled her interest in conservation and sustainable development issues. After three years of working with a land stewardship initiative in Ann Arbor, Michigan, she joined Peace Corps Paraguay in 1998. She remained in Paraguay for three years, working first as an agroforestry volunteer with a farmer's organization and later as the coordinator for the Peace Corps Paraguay environment program. Upon returning to the US in 2002, she interned for Conservation International's Conservation Coffee program in Washington, DC. In 2003, she began her graduate program at University of Florida in the School of Forest Resources and Conservation and with the Tropical Conservation Development program. She conducted her master's research in 2004 and 2005 in Acre, Brazil, investigating the impacts of a community-based timber operation on future crop trees. She plans to continue her doctoral work at the University of Florida.