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THE IMPACT OF FSC CERTIFICATION ON TIMBER TREE REGENERATION AND FLORISTIC COMPOSITION IN HONDURAN COMMUNITY FORESTS

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ABSTRACT

Forest certification has been suggested to provide a promising means to improve the sustainability of forest management in the tropical countries, where traditional environmental regulation has been inefficient in controlling forest degradation and deforestation. In these countries, communities have an increasingly important role as managers of tropical forest resources. However, thus far only a fraction of community forests have been certified, and little remains known about the suitability of certification in improving the sustainable management of community-based forest operations.

Two areas in Honduras where community-managed forest operations had received certificates of good forest management were studied. Río Cangrejal represents an area with longer use history, whereas Copén is a more recent forest operation on the frontier of a biosphere reserve. The aim of the study was to assess the ecological sustainability of forest management through comparing timber tree regeneration and the vegetation composition between certified, conventionally managed and natural forests. Data on woody vegetation and environmental conditions was collected in logging gaps and natural treefall gaps.

The regeneration success of the shade-tolerant timber tree species was lower in certified than in conventionally managed forest logging gaps in Río Cangrejal, although the environmental conditions indicated reduced logging disturbance in the certified forests. Furthermore, the vegetation composition was more natural-like in the conventionally managed than the certified forests. It was suggested that the certified forests may have been subjected to more intensive pre-certification loggings, whereas the post-logging recovery of the conventionally managed forests may benefit from their closer proximity to protected forest areas.

The results in Copén demonstrated that the regeneration success of light-demanding timber species was higher in certified than unlogged forests. However, the regeneration of the commercially valuable *Swietenia macrophylla* may be hampered due to suppression from competing vegetation. The better accessibility of the unlogged forests may partly explain the lack of economically valuable species.

The results were analysed together with results of a socioeconomic study, to identify the challenges of forest certification in community-based forestry in the tropics. Four factors were identified that need increased attention to make FSC certification a more efficient instrument of sustainable forest management in community operations:

1) The ecological as well as the social systems are heterogeneous. Evaluating the impacts of logging is demanding, and management guidelines aimed at reducing the mechanical logging damage may favour some timber species over others. Better recognition of the diverse roles of forestry in the livelihood strategies of the various community stakeholders is needed.

2) The socio-ecological landscapes are modified by past resource use histories. Degraded forests may be fragmented to a degree, further impeding the post-logging recovery of the

ecosystems. Actions to improve forest recovery are linked to better recognition of resource rights.

3) In these types of landscapes, successful implementation of ecologically sustainable management should involve the agropastoral areas between forests. Similarly, focus should be expanded over the community level to improve the overall status of community producers and specifically to distribute responsibilities in controlling illegal activities to governmental authorities.

4) Ultimately, the feasibility of forest certification is dependent on its ability to transform the existing market structures for certified forest products to become more beneficial for community forest producers.

Key words: community forest management, forest certification, Forest Stewardship Council, Honduras, natural regeneration, sustainable forest management, reduced-impact logging (RIL)

LIST OF ORIGINAL PUBLICATIONS

This thesis is based on the following publications, which are referred to in the text by their Roman numerals:

I Kukkonen, M., Rita, H., Hohnwald, S., Nygren, A. 2008. Treefall gaps of certified, conventionally managed and natural forests as regeneration sites for Neotropical timber trees in northern Honduras. Forest Ecology and Management 255: 2163-2176.

II Kukkonen, M., Hohwald, S. 2009. Comparing floristic composition in treefall gaps of certified, conventionally managed and natural forests of northern Honduras. Annals of Forest Science 66: 809. DOI: 10.1051/forest/2009070.

III Kukkonen, M., 2010. The impact of community-based forest management on timber tree regeneration in north-eastern Honduras, with specific reference to Big-Leaf Mahogany (*Swietenia macrophylla*). Southern Forests: a Journal of Forest Science 72 (3/4): 133-140.

IV Bieri, M., Nygren A. 2011. The challenges of certifying community forestry in the tropics: a case study from Honduras. Journal of Environment & Development (available online, doi:10.1177/1070496511405154).

Author's contribution:

I The original idea was developed by SH and AN. SH collected the main field data, while MB collected additional field data. HR planned the approach and statistical analyses together with MB. MB conducted the statistical analyses and was responsible for the writing process together with HR and in collaboration with the other co-authors.

II The original idea was developed by SH and AN. SH collected the main field data, while MB collected additional data. MB was mainly responsible for planning the statistical analyses, and conducted all the analyses. MB was responsible for the writing process, with a contribution from SH.

III The original idea was developed by MB but based on earlier work developed by SH and AN. MB collected the data, conducted the analysis and wrote the article with a major contribution from HR.

IV The original idea was developed by AN and MB. MB contributed to the environmental part of the article and AN to the socio-economic part of the article. The working load in the planning and writing process was distributed equally among the authors. MB acted as the corresponding author.

MB = Mari Bieri (Kukkonen), HR = Hannu Rita, SH = Stefan Hohnwald, AN = Anja Nygren

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LIST OF ABBREVIATIONS

AFE-COHDEFOR	Administración Forestal del Estado – Corporación Hondureña de Desarrollo Forestal (State Forestry Administration – Honduran Forestry Development Corporation)
CATIE	Centro Agronómico Tropical de Investigación y Enseñanza (Tropical Agronomy Teaching and Research Centre in Costa Rica)
CeF	Certified forest
COATLAHL	Cooperativa Regional Agro-Forestal Colón, Atlántida, Honduras, Ltda. (Regional Agroforestry Cooperative of Colón and Atlántida, Honduras, Ltd.)
CoM	Conventionally managed forest
Dbh	Diameter at-breast-height (130 cm)
DDCA	Detrended Correspondence Analysis
ESNACIFOR	Escuela Nacional de Ciencias Forestales, Honduras (National School of Forest Sciences)
FAO	United Nations Food and Agriculture Organisation
FMU	Forest management unit
FS	Floristic similarity
FSC	Forest Stewardship Council
GEE	Generalized estimating equations
GF	Gap favourability
HCVF	High Conservation Value Forest
ITTO	International Tropical Timber Organisation
MN	Management-neutral
MS	Management-sensitive
NaF	Natural (here: protected/unlogged) forest
NGO	Non-governmental organisation
NTFP	Non-timber forest product
PCA	Principal Components Analysis

PEFC	The Program for the Endorsement of Forest Certification
pRDA	Partial Redundancy Analysis
RDA	Redundancy Analysis
RIL	Reduced-impact logging
RPBR	Río Plátano Biosphere Reserve
RS	Regeneration success
SERNA	Secretaría de Recursos Naturales y Ambiente
SFM	Sustainable forest management
STY	Sustained timber yield
UNECE	United Nations Economic Commission for Europe

1. INTRODUCTION

1.1 SELECTIVE LOGGING IN THE HUMID TROPICAL FORESTS

The humid tropical forest zone includes the wet, moist and semi-deciduous forest types. It covers about 85% of the world's 1.68 million ha of tropical forests (Dupuy et al., 1999). The humid tropical forests have global importance as reservoirs of the majority of terrestrial species diversity (Dirzo and Raven, 2003). They are typically characterized by a very high richness of tree species; over 280 species (dbh ≥ 10 cm) have been recorded in a single hectare of Amazonian forest (Valencia et al., 1994; De Oliveira and Mori, 1999). Selective logging, in which the valuable species are selectively removed from the forest, is the dominant form of timber harvesting in these forests. Selectively logged forests are usually classified as modified natural forests; although they are composed of naturally regenerating native species, logging has affected their structure and species composition (FAO, 2006).

The humid tropical forests have been subjected to rapid deforestation, which continues to date: between the years 2000 and 2005, the area of humid tropical forest was reduced by an estimated 2.4% (Hansen et al., 2008). The main cause of deforestation throughout the region is the conversion of forests to agriculture and cattle pasture (Achard et al., 2002). Agricultural conversion is often facilitated by selective logging, which improves access to the forests due to the construction of roads and wood transport routes (Dupuy et al., 1999). Furthermore, selective logging has caused ecological degradation in large tracts of forest; according to an ITTO (2006) estimate, only about 5% of the natural tropical forests are sustainably managed. Forest governance institutions in the tropical countries are often underfunded and corrupt, and the control of illegal logging is inefficient (ITTO, 2006). The lack of resources limits the efficient regulation of harvesting rates and techniques (ITTO, 2006; FAO, 2009). Although increasing urbanization is expected to relieve the pressure of the conversion of humid tropical forests to agricultural purposes, the deforestation and degradation of humid tropical forests continue, due to a lack of efficient instruments of sustainable management (FAO, 2009).

1.1.1 ECOLOGICAL IMPACTS OF SELECTIVE LOGGING

Although the ecological impacts of selective logging are less pronounced than those of clear-felling, the changes in forest structure and species composition may be profound. Perhaps the most common way of assessing the ecological impacts of selective logging is through measuring changes in the post-logging regeneration of timber trees; this approach is used in papers I and III. Generally, selective logging affects timber tree regeneration in two ways: through limiting pollination and seed dispersal, and through affecting the distribution of suitable regeneration niches. The relative importance of these two processes varies in space and time, as well as according to the tree species.

The spatial distribution of seed trees is the main determinant of species composition in humid tropical forests (Dalling et al., 1998; Fredericksen and Licona, 2000; Grau, 2004; Hubbell and Foster, 1986; Webb, 1998). Exhaustive selective harvesting reduces the abundance of the timber tree species, sometimes up to the point of local extinction (Fredericksen and Licona, 2000; Hall et al., 2003; Lindenmayer et al., 2000). Many economically important timber species have large seeds with short dispersal distances and short viability times, which makes them particularly vulnerable to the local impacts of selective logging (Wijdeven and Kuzee, 2000). Furthermore, many of the tree species of the humid tropical forests are rare; the recorded proportion of species occurring at a density of less than one individual per hectare was a third of all species in Panama (Hubbell and Foster, 1986) and nearly half in Cameroon (Kenfack et al., 2007). The small population size increases the vulnerability of these species to local extinctions. Dioecious species have been found particularly vulnerable to logging, due to the inability of pollinator species to cover the increasing distances between tree individuals (Mack, 1997).

In the long term, selective logging may affect the viability of the timber tree populations through reduced genetic variation. This happens when the rarest alleles are removed from the population by harvesting. Furthermore, logging typically focuses on the tallest, straightest growth forms, which results in dysgenic selection, i.e. a gradual increase in the relative abundance of the poorly formed genotypes (Jennings et al., 2001). Gillies et al. (1999) sampled *Swietenia macrophylla* King (Meliaceae) populations in Central America, finding that those populations with the longest history of exploitation exhibited lower genetic diversity than the unlogged populations.

Selective logging also affects tree regeneration through changing the distribution of microsites that function as regeneration sites. The canopy of humid tropical forests is typically thick and multi-layered, and the main factor limiting growth in the understory is light. Therefore, disturbances that create canopy openings have a particularly important role in tree regeneration dynamics (Augspurger, 1984; Denslow, 1987; Dupuy and Chazdon, 2006; Fraver et al., 1998; Oldeman and van Dijk, 1991). These disturbances vary in size from the falling of a single branch to the burning of a whole forest – and in time, from slow-progressing erosion to a sudden storm (e.g. Perry and Amaranthus, 1997). In selectively logged forests, the frequency and size distribution of the canopy gaps differs from natural, unlogged forests. The felling of trees, together with the construction of logging roads and skid rails, results in the loss of canopy cover (Bawa and Seidler, 1998; Jackson et al., 2002; Uhl and Vieira, 1989; White, 1994).

The loss of canopy cover changes the relative abundances of the plant species, because an ecological trade-off exists between a species' ability to benefit from increased light and its tolerance of shaded conditions (e.g. Baraloto et al., 2005). This trade-off is used in the ecological classification of tropical trees into light-demanding and shade-tolerant species. The light-demanding (i.e. shade-intolerant) species thrive in large canopy openings (Brokaw and Scheiner, 1989). They have a good ability to react to increased light with increased growth (Popma and Bongers, 1988). The light-demanding species typically possess a pioneer-type life strategy, i.e. they produce a large number of small, wind-dispersed seeds, with which they may disperse efficiently (Foster and Janson, 1985; Whitmore, 1989). The shade-tolerant (i.e. non-pioneer or light-independent) species, on the contrary, are able to germinate under a closed canopy (Brokaw, 1985, Swaine and Whitmore, 1988; Whitmore, 1989). These species have a limited ability to react to increased light availability with improved growth (Popma and Bongers, 1988); however, they may be more resistant to damage caused by pathogens (Augspurger and Kelly, 1984) or herbivores (Blundell and Peart, 2001) than the light-demanding species. The shade-tolerant species generally possess fewer and larger seeds, and are longer-lived, compared to the light-demanding species (Foster and Janson, 1985; Whitmore, 1989). Although the ecological requirements of the tree species rarely completely fit in either group (e.g. Brokaw, 1987; Denslow, 1980; Swaine and Whitmore, 1988; Whitmore, 1989), this classification has been widely used in tropical forest ecology, and it is used in papers I, II, and III. Further groupings, based on the life-history characteristics of the species, are often made (e.g. Poorter et al., 2006).

The logging-induced changes in forest light conditions typically favour the light-demanding over the shade-tolerant species. Depending on the intensity and implementation of loggings (Bawa and Seidler, 1998), the mechanical disturbance of felling and transporting wood may damage and destroy trees over a relatively large area (Asner et al., 2004; Cannon et al., 1994; Dickinson et al., 2000; Feldpausch et al., 2005; Hall et al., 2003; Jackson et al., 2002; Johns, 1988; Uhl and Vieira, 1989; Whitman et al., 1997; Woods, 1989). The shade-tolerant species are particularly vulnerable to this type of damage, due to being reliant on advance regeneration (i.e. their seeds germinate under closed canopy and may persist in low light conditions for several years) (Felton et al., 2006). The light-demanding species, on the other hand, are typically stronger competitors in high light conditions. Primack and Lee (1991) found the abundance of light-demanding trees to increase significantly as a result of selective logging in the Bornean rainforests. Dickinson et al. (2000) recorded four times more stems of light-demanding species in skidder-disturbed logging gaps compared to natural treefall gaps in Mexico. Hall et al. (2003) observed a lower basal area of shade-tolerant trees in post-logged forests compared to unlogged areas in Central Africa. In addition to trees, improved light availability has been found to increase the relative abundance of light-demanding lianas (Schnitzer and Bongers, 2002; Schnitzer et al., 2004), herbs, shrubs (Babaasa et al., 2004;

Chapman and Chapman, 1997) and bamboos (Tabarelli and Mantovani, 2000) in selectively logged forests. The impacts of selective logging may persist in the forest structure and species composition for decades (Okuda et al., 2003).

Besides light conditions, selective logging also affects soil texture and microclimate. Soil compaction (Whitman et al., 1997) and the loss of organic matter and nutrients may limit the early growth of trees, particularly when heavy machinery is used in harvesting (Nussbaum et al., 1995). McNabb et al. (1997) found the changes in soil pH, bulk density and nutrient concentrations to persist for up to 16 years after logging. Increased light influx elevates the ground temperatures and causes drought in the understory, making forests more vulnerable to fires (Ray et al., 2005). In a study conducted in eastern Amazonia, Holdsworth and Uhl (1997) found that the susceptibility to fire increases in relation to the size of the logging gap.

The impacts of selective logging may affect the fluxes of carbon, water and nutrients in the long term; with the shift towards herbaceous species and younger trees, the plant community is not able to utilize water from the deeper soil layers, which limits canopy moisture and greening during the dry season (Koltunov et al., 2009). Furthermore, the changes in forest structure and floristic composition are reflected as changes in forest fauna and in the interactions between organisms (Dirzo and Miranda, 1990). Lambert et al. (2005) found increased abundances of small mammal species and increased seed predation rates in logged forests of southeastern Amazon. Even low-intensity logging has been found to cause homogenization of biodiversity (Bawa and Seidler, 1998; Hamer and Hill, 2000). Generally, generalist species are favoured over primary forest specialist species (Thiollay, 1997).

Especially in the case of tropical forests, the indirect impacts of selective logging may often be more detrimental to the ecosystems than the direct impacts. The building of transport routes for selectively logged timber opens new access to the forests, which increases the risk of agricultural conversion. The improved access to forests may also increase the intensity of hunting and collection (Thiollay, 1997; Wright, 2003). Due to the importance of vertebrate seed dispersal agents, hunting may also negatively affect the regeneration of timber tree species (Nuñez-Iturri and Howe, 2007; Wright, 2003).

1.1.2 Reduced-impact logging

A range of reduced-impact logging (RIL) guidelines have been developed to control the negative ecological impacts of selective harvesting. RIL techniques include improved pre-harvest planning, such as the inventorying of crop trees and the setting of logging quotas and minimum logging diameters to sustain the populations of the timber species (Dykstra, 2001). Mechanical disturbance is minimized by controlling the construction of roads and skid trails (Bertault and Sist, 1997). Liana cutting from the felled trees prior to logging is practiced to prevent the falling of adjacent trees (Schnitzer et al., 2004) and to reduce the proliferation of lianas in the logging gaps (Gerwing and Uhl, 2002). Directional felling is practiced to minimize the logging damage on future crop trees and residual vegetation (Bertault and Sist, 1997). Further techniques are related to reducing wasted wood and avoiding damage to vegetation during log handling. Post-harvest assessments are required to evaluate the ecological impact of logging (Dykstra, 2001).

Several studies have shown that the ecological impacts of selective logging can be reduced by using RIL. The ground-level disturbance per felled tree (Asner et al., 2004; Pereira et al., 2002) and the damage caused to the residual stand (Bertault and Sist, 1997) were found to be lower in RIL compared to conventionally logged tropical forests. The logging gaps in RIL are generally smaller and close faster (Asner et al., 2004), which may reduce the susceptibility to fire (Holdsworth and Uhl, 1997). Putz et al. (2008) found loggers using RIL to waste less wood. Furthermore, the results of Olander et al. (2005) suggest that RIL may help to reduce the changes in soil composition. Davis (2000) reported RIL forests to host more natural-like assemblages of beetle species compared to conventionally logged forests. Although the wider-level impacts are generally poorly known, the results of Feldpausch et al. (2005) in Amazonia suggest that RIL forests may store more carbon than conventionally logged forests.

1.1.3 Ecological sustainability in selectively logged humid tropical forests

Sustainable forest management (SFM) has three dimensions: social, ecological and economical sustainability. It is closely related to the concept of sustainable development, which means guaranteeing the needs of the current generation without compromising those of future generations (Callicott and Mumford, 1997). The concept of ecologically SFM was developed to replace the earlier view of good forest management as sustained timber yield (STY) and the preservation of 'wilderness areas' for the protection of native species (Callicott and Mumford, 1997). Ecological sustainability relies on the recognition of the role of human-managed areas in biodiversity conservation (e.g. Fredericksen and Putz, 2003). Although ecological sustainability is an ambiguous concept with varying definitions, some commonly agreed standards can be identified. These include the maintenance of a pristine forest species composition, structure, biodiversity, and ecosystem functions. However, no thresholds exist for the accepted levels of change, which makes it impossible to unambiguously define whether a management system fulfils the requirements of ecological sustainability.

In tropical forestry, the use of RIL is a core requirement in all schemes that promote ecologically sustainable management. However, the central role of RIL in SFM has been criticized in many ways. First, RIL is mainly focused in timber production, while little attention is paid to the functions of the forests in providing various non-timber forest products (NTFPs), such as hunting or the collection of plants for food and medicine, and environmental services (García-Fernández et al., 2008; Ros-Tonen et al., 2008). Second, whilst RIL methods have been practiced in the northern hemisphere with good success, the requirements for training and financial investments may exceed the resources available in the developing countries (Pokorny et al., 2005).

Third, RIL does not necessarily guarantee STY, but may even harm the regeneration of some timber tree species in tropical forests. Minimizing logging disturbance may create logging gaps that are too small to allow the regeneration of light-demanding timber tree species (Brokaw and Scheiner, 1989; Dickinson et al., 2000; Snook and Negreros-Castillo, 2004; Webb, 1998; III). The use of silvicultural measures that have been applied to successfully improve the regeneration of light-demanding timber tree species in tropical forests (e.g. Hartshorn, 1989; Peña-Claros et al., 2008) is incompatible with the RIL aim of minimizing the ecological impacts of logging (IV). Fourth, RIL is typically implemented with set standards, for instance, for the proportion of seed trees and minimum logging diameters. In tropical regions, available information on species-specific phenological traits and ecological requirements is often insufficient for the planning of ecologically feasible guidelines (Grogan and Landis, 2009; Guariguata and Pinard, 1998; Hartshorn, 1995); in addition, there are often no case-specific data available on the population structure and distribution of each species (Freitas and Pinard, 2008; Schulze et al., 2008).

1.2 FOREST CERTIFICATION

Forest certification is a market-based instrument for promoting sustainable forest management. The aim of forest certification is to provide an assurance of sustainable production to the consumer through product labelling. There are two types of certifications. In forest management certifications, the management system is evaluated against a set of standards that consider the different aspects of sustainability. Chain-of-custody certifications aim at securing that the source of the certified material is tracked through the transport and processing of the end product (Elliott, 2000).

1.2.1 ORIGINS OF FOREST CERTIFICATION

The creation of forest certification in the early 1990s reflected the widespread concern over the rapid deforestation and degradation of tropical forests. Traditional environmental regulation was seen to produce insufficient measures to protect the tropical forests, and the attempts to create a global forestry convention at the Rio de Janeiro Earth Summit in 1992 had failed (Van Kooten et al., 2005). Boycotts and bans on tropical timber trade were proving to be poor tools for conserving the forests; in fact, the lowered value of timber often led to the conversion of forests to other land uses (Nussbaum and Simula, 2005). At the same time, the relationship between trade and the environment was high on the international policy agenda (Elliott, 2000). Market-based instruments were increasingly seen as credible alternatives to traditional government-based environmental regulation. Furthermore, negative campaigning by non-governmental organisations (NGOs) had made large retailer companies receptive to the idea of a labelling system that would help them show to the public that their products had been produced in an environmentally and socially acceptable manner (Nussbaum and Simula, 2005). The environmental NGOs were also starting to see the potential value of managed forests in the global efforts to conserve biodiversity.

The Forest Stewardship Council (FSC) was officially established in 1993 as the first forest certification scheme. It was formed by international environmental NGOs working together with indigenous peoples' groups and industry representatives (Klooster, 2005). Following the creation of the FSC, other certification schemes were soon established, many of them with support from the forest industry rather than environmental NGOs (Rametsteiner and Simula, 2003).

1.2.2 Status and Trends of Forest Certification

In 2010, the area of certified forests covered 355 million hectares, about 9% of the world's total forest area (UNECE/FAO, 2010). Since the early years of certification, the area under certified forestry has been expanding most rapidly in the developed countries of the Northern hemisphere, where over 80% of the certified forest area is currently situated. In Africa, Asia and Latin America, less than 2% of the total forest area is certified, whereas in Western Europe, more than half of the forests have been certified (UNECE/FAO, 2010). Besides this geographical trend, it is also clear that the expansion has favoured large-scale over small-scale forest operations. Among Southern producers, community-based forestry enterprises form a particularly poorly represented group in certification. Only 1% of community forests worldwide had been certified in 2002, and this portion was considered unlikely to reach more than 2% within the next twenty years (Molnar, 2003).

The lack of participation by Southern producers in certification may be partly explained by the lack of price premiums for certified wood. Although such premiums were initially intended as a part of the certification system, they have not been realized in most cases. Southern producers also face difficulties in coping with the direct and indirect costs of certification (Klingberg, 2003). In the non-environmentally sensitive markets, certified wood that is produced with higher costs has to compete with illegally logged timber products (Gullison, 1998). Another reason for the dominance of Northern producers is that they are better able to produce large quantities of even-quality wood, and hence gain a stronger foothold on the market (Taylor, 2005a).

The rapid expansion of the certified forest area is largely due to growing environmental awareness and corporate responsibility programmes (Taylor, 2005a; Overdevest and Rickenbach, 2006). In the future, the global certified forest area is expected to continue growing, driven by initiatives such as green building standards (Overdevest and Rickenbach, 2006; UNECE/FAO, 2008). The markets for certified products have so far been limited to North America and Western Europe. In the future, the main consumers of many wood products will be in the non-environmentally-sensitive markets of Asia, although the highest per-capita consumption rates remain in Europe and North America (FAO, 2009). Nevertheless, certification has been seen as an important mechanism for promoting SFM in tropical forests (Brown et al., 2001). FAO (2009) listed the limited expansion of certification as one of the main obstacles to SFM in the tropics, where traditional environmental regulation has been largely unsuccessful in hindering the rapid deforestation and degradation of forests.

1.2.3 FSC CERTIFICATION

This study focuses on Forest Stewardship Council (FSC) forest certification. Along with PEFC (the Program for the Endorsement of Forest Certification, initially Pan-European Forest Certification), FSC is the other major global certification system. By January 2010, FSC had certified over 120 million ha of forestland and plantations in 81 countries around the world (FSC, 2010). Out of the existing international certification schemes, FSC is the most prevalent in the tropical areas. Over half of all certifications in tropical forest areas are under the FSC scheme (UNECE/FAO 2008).

FSC is an independent, non-profit certifying organisation that functions by maintaining and promoting a set of performance-based principles and criteria of forest management (Elliott, 2000). The ten general principles and their sub-criteria are used as a basis for evaluating the sustainability of forest management (Table 1); in paper IV, the feasibility of FSC certification in improving sustainability of forest management in the studied Honduran forests is evaluated against these principles. FSC works by accrediting organisations that grant certifications, using the FSC principles and criteria as a basis in evaluating forest management. The accredited organisations are also expected to develop area-specific criteria, which are based on the FSC standards.

Table 1. The FSC principles with their main requirements (FSC, 2002).

Principle	Main requirements
#1: Compliance with laws and FSC Principles	• Forest management shall respect all national and local laws, and relevant international treaties and agreements, and comply with all FSC principles and criteria.
#2: Tenure and use rights and responsibilities	• Long-term tenure and use rights to the land and forest resources shall be clearly defined, documented and legally established
#3: Indigenous people's rights	• The legal and customary rights of indigenous peoples to own, use and manage their lands, territories, and resources shall be recognized and respected.
#4: Community relations and workers' rights	• Forest management operations shall maintain or enhance the long-term social and economic well-being of forest workers and local communities.
#5: Benefits from the forest	• Forest management operations shall encourage the efficient use of the forest's multiple products and services to ensure economic viability and a wide range of environmental and social benefits.
#6: Environmental impact	• Forest management shall conserve biological diversity and its associational values, water resources, soils, and unique and fragile ecosystems and landscapes, and by so doing maintain the ecological functions and the integrity of the forest.
#7: Management plan	• A management plan – appropriate to the scale and intensity of the operations – shall be written, implemented, and kept up to date. The long-term objectives of management, and the means of achieving them, shall be clearly stated.
#8: Monitoring and assessment	• Monitoring shall be conducted – appropriate to the scale and intensity of forest management – to assess the condition of the forest, yields of forest products, chain of custody, management activities and their social and environmental impacts.
#9: Maintenance of high conservation value forests	• Management activities in high conservation value forests shall maintain or enhance the attributes which define such forests. Decisions regarding high conservation value forests shall always be considered in the context of a precautionary approach.
#10: Plantations	• Plantations should reduce the pressures on, and promote the restoration and conservation of natural forests. They should be designed in a way to enhance the conservation of biological diversity. Natural species should be preferred over exotics.

FSC offers a good target for studying the impacts of certification, because its environmental standards are generally thought to be the most rigorous of the existing certification schemes (Klooster, 2005). FSC's values of sustainability and social responsibility reflect the strong role of NGOs (Klingberg, 2003). FSC is governed by three chambers: the economic, environmental and social chamber. The purpose of this decision-making structure is to prevent the dominance of specific interests. Each of these chambers has representatives from the South as well as from the North.

The FSC certification process starts with the initiative of the forest owner or manager who applies for certification. The certifier organisation, which is accredited by FSC, conducts a preliminary assessment, producing a report for the forest manager. After this, the official evaluations are started; a team of experts from different fields conducts the assessments by conducting field inspections, evaluating necessary documents including management plans and forest inventories, and consulting stakeholders (Elliott, 2000). After the issuing of a certificate, the process continues with annual follow-up audits. Minor non-compliance with FSC standards may lead to the setting of corrective actions (Nussbaum and Simula, 2005).

Different classes of certified forestry are used according to how well the certified operation fulfils the FSC criteria. SmartWood, which is one of the main certifying organisations of the FSC, issues certifications of 'sustainable' and 'well-managed' forestry. A forest operation that is certified as sustainable has follow-up data to prove sustainability in the long term (Vogt and Fanzeres, 2000). A certificate of good management may be given to a forest operation that shows less strict commitment to the given criteria and lacks long-term data (Higman et al., 2005). The third class of certified management, 'pre-certified', implies that the operation needs to fulfil certain improvements to become certified.

1.2.5 The Impact of Forest Certification in Global Forest Conservation

The success of forest certification in promoting SFM in the world's forests can be evaluated in several different ways. Perhaps the clearest measure is the worldwide coverage of the certified forest area. The geographical bias towards the Northern Hemisphere limits the global effectiveness of certification in two main ways. First, tropical forests have particular importance in the conservation of global terrestrial biodiversity, and in mitigating climate change. Second, the improvements related to certification are dependent on whether certification will be able to reach those forests that are currently most threatened by degradation or deforestation. These areas are largely in the tropical region, whereas the forests in the developed countries of the Northern

Hemisphere often already fill many of the certification requirements, due to stringent national regulations and good institutional capacity (Gullison, 2003; Simula, 1999).

The effectiveness of certification in securing SFM can also be measured through the feasibility and relevance of the certification criteria. Evidence shows that the certifying of forest management systems has changed forestry practices in many areas (Auld et al., 2008; Newsom et al., 2006). However, the impacts also depend on the certification scheme, as the stringency of criteria for the different aspects of SFM varies among the different certifiers (Auld et al., 2008). According to Vogt et al. (2000), the competition between certification schemes and the rapid expansion of the certified forest area have led to the development of criteria that are too general and allow a variety of interpretations on what constitutes sustainable management in local conditions. Especially in the tropical areas, the feasibility of certification in improving SFM is also limited by difficulties in resolving the trade-offs between the ecological, social and economic criteria (IV).

1.2.6 THE CERTIFICATION OF COMMUNITY OPERATIONS IN HUMID TROPICAL FORESTS

The role of communities as managers of the tropical forest resource is rapidly increasing in importance. According to an estimate by White and Martin (2002), almost a quarter of the forest area in the 18 most forested developing countries was owned or managed by local communities in the early 2000s. Due to decentralization policies and the devolution of forest resources to local communities, the share of the forest area under community management increased by 26% between 2002 and 2008 in the 25 most forested countries (Sunderlin et al., 2008). Generally, increased community ownership of natural resources is thought to decrease deforestation rates. Supporting this assumption, a study by Ellis and Porter-Bolland (2008) in Mexico showed that community forest management may be a better way to prevent deforestation than setting up conservation areas.

Considering the importance that communities hold in the conservation of tropical forests, and the potential of forest certification to enhance sustainability in those areas where forest policies are insufficient to prevent the degradation of forests, there have been surprisingly few studies assessing the feasibility of forest certification as an instrument of improved sustainability in community-management systems. This may be partly due to the seemingly low ecological impact of community-based harvesting systems. However, community operations have several characteristics that may limit the success of reaching SFM through certified forestry. The ecological impacts of logging are difficult to predict in the poorly known ecosystems (Nussbaum and Simula, 2005). The various community stakeholders may have different interests towards the forest resource. Furthermore, the unsupportive economic and political structures may significantly limit the success of efforts to increase the sustainability of harvesting (Ebeling and Yasué, 2009; van Kooten et al., 2005). The communities that live in or near the forests in the developing countries are often dependent in the forest products and ecosystem services in many ways; hence, the successful integration of the social and ecological aspects of forest management is particularly crucial in these systems (IV).

2. OBJECTIVES AND HYPOTHESES

2.1 RESEARCH OBJECTIVES

The main aim of this study was to assess the impact of certified management practices on the regeneration of timber tree species (I, III) and the changes in natural forest (NaF) species composition (II) in community-based forest operations in Honduran tropical moist forests. This approach was chosen because sustaining the regeneration of timber tree species to compensate for their removal by logging and minimizing changes in the natural forest structure and species composition can be considered to be two main aims of ecologically sustainable forest management (Vogt et al., 2000).

In earlier studies, the impact of improved forest management practices has usually been evaluated as either 1) decreased damage to residual trees (Bertault and Sist, 1997; Johns et al., 1996; Holmes et al., 2002), 2) a decreased area affected by tree felling, log landings, skid rails and roads (Asner et al., 2004; Boltz et al., 2003; Johns et al., 1996; Holmes et al., 2002; Huth et al., 2004; Pereira et al., 2002), or 3) changes in the structure or species composition of the tree community (Boltz et al., 2003; Huth et al., 2004; Pereira et al., 2002). In these studies, it has been assumed that the actions taken to reduce the environmental impact of logging result in enhanced regeneration of the logged timber species and help to minimize the changes in natural species composition. In this study, particular interest was paid to studying *how* the improved harvesting methods affect the environmental conditions within canopy gaps, and their favourability as regeneration sites for timber species (I, III) and in sustaining a natural-like species composition (II). To do this, data were collected on the species composition and environmental gap conditions within the logging gaps of certified forests (CeFs), and compared to data collected from natural treefall gaps of NaFs and logging gaps of conventionally managed forests (CoMs), where possible (I, II, III).

A further aim of the studies forming this thesis was to identify the challenges of FSC certification in improving the sustainability of tropical community-based forest management (IV). FSC forest certification aims at promoting forest management that is “environmentally appropriate, socially beneficial and economically viable” (FSC, 2009).

These three aspects of SFM are deeply interlinked, and the linkages should be taken into account when formulating practical recommendations for the improvement of the FSC criteria. Therefore, for this part of the work, a multidisciplinary approach was taken; the results of the ecological study were assessed together with those of a socioeconomic study conducted in the same area (IV).

2.2 HYPOTHESES

2.2.1. THE REGENERATION SUCCESS OF TIMBER SPECIES

The certification criteria specify several actions to be taken in CeFs, such as the carrying out of preharvest inventories and the using of minimum logging diameters, to guarantee a sustained yield of the logged timber species. Certification requirements emphasize the minimizing of mechanical logging damage, which may be expected to favour the shade-tolerant timber tree species over the light-demanding species (see Chapter 1.2.2) (I). Hence, the first hypothesis on timber regeneration success was defined as follows:

RS(a) The regeneration success of shade-tolerant timber species in CeF gaps is higher than in CoM gaps (I).

Although the impact of logging is minimized in CeFs, there are still several aspects in which CeFs differ from NaFs. When systematic compensatory planting of timber trees is not required, the removal of mature trees in CeFs limits the relative abundance of seed sources for these species. In addition, even low-impact logging causes some level of damage to the developing trees (I). In managed forests, the larger size of the canopy openings and the mechanical removal of competing vegetation may be expected to benefit the regeneration success of the light-demanding timber species relative to the shade-tolerant timber species (III). Based on this, the following hypotheses on regeneration success were set:

RS(b) The regeneration success of shade-tolerant timber species in CeF gaps is lower than in NaF gaps (I, III).

RS(c) The regeneration success of light-demanding timber tree species is higher in CeF than NaF gaps (III).

By reducing mechanical logging damage, certified forestry aims at controlling the damage caused to the regenerating timber tree species, and at preventing the dominance of secondary species. Hence, there should be less logging-related damage in CeF gaps compared to CoM gaps. However, even when the disturbance in CeFs is controlled, NaFs represent ‘minimum disturbance’ environments. Hypotheses on gap favourability were based on these considerations:

GF(a) CeF gaps create more favourable environments for shade-tolerant timber regeneration than CoM gaps (I).

GF(b) CeF gaps create less favourable environments for shade-tolerant timber regeneration than NaF gaps (I, III).

GF(c) CeF gaps create more favourable environments for light-demanding timber regeneration than NaF gaps (III).

2.2.2 THE NATURALNESS OF THE FLORISTIC COMPOSITION

Along with sustaining the populations of timber tree species, the maintenance of NaF species composition is a main aim of certified forest management. The impacts of logging on biodiversity are controlled in certified forestry by restricting the area of logged segments, designating set-asides for the protection of vulnerable species and ecosystems, and by limiting the harvesting intensities and the overall mechanical logging damage (II). Based on these considerations, the following hypotheses on floristic similarity and the favourability of CeF gaps to NaF species composition were formulated:

FS: The floristic composition is more similar between CeF and NaF gaps than between CoM and NaF gaps (II).

GF(d): The environmental conditions in CeF gaps support a more natural-like species composition than the conditions in CoM gaps (II).

3. MATERIAL AND METHODS

3.1 HONDURAN FORESTRY

The Honduran climate is humid and warm by the Atlantic coast, and cooler and drier in the central highlands (SERNA, 2001). The country is highly vulnerable to hurricanes and cyclones, typically moving from east or southeast through the Caribbean Sea (Wadsworth, 1997). The impacts of hurricanes have been particularly devastating for the rural poor (Guill and Shandera, 2001). About half of the Honduran population of 7.2 million people are considered to live below the national poverty line, and 30% earn less than two USD a day. In 2008, about 52% of the Honduran population were rural, although this proportion is steadily decreasing (World Bank, 2009). The Honduran economy is largely dependent on a few export products, such as bananas and coffee (CIA World Factbook, 2008). About a third of the Honduran population are employed in agriculture. However, due to the steep slopes and poor soils, large areas in Honduras are unsuitable for agricultural purposes (Merrill, 1995).

In 2005, 42% of the Honduran land area of 11.2 million hectares was covered by forests (World Bank, 2009). About half of the forested area is tropical moist broadleaved forest, which spreads across the northern Atlantic coast and throughout the eastern Mosquitian lowlands. The other half mainly consists of pine-dominated forests of the central highlands. Between 2000 and 2005, the yearly deforestation rate in Honduras was 3.1% − the highest out of the Central American countries, and amongst the 10 highest relative deforestation rates in the world (FAO, 2006). Main causes of deforestation are the conversion of land for agriculture, illegal logging and corruption of forestry authorities (Richards et al., 2003). Whereas commercial forest production mainly focuses on *Pinus* species, illegal logging is more prevalent in broadleaved forests. This is partly explained by the fact that a far higher proportion of pine forests are covered by management plans compared to broadleaved forests (ITTO, 2006). In practice, all logging that is not executed according to the approved forest management criteria and permits is classified as illegal. However, not all of these operations count as clandestine production; a major part of unregulated forestry production is registered and fraudulently legalized (Del Gatto, 2004). Against this background, it is rather alarming that the estimated proportion of legal and legalized logging together is merely a quarter of the total hardwood production in Honduras, while some 75–85% of all hardwood is produced by clandestine operators (Richards et al., 2003).

The large-scale commercial exploitation of Honduran hardwoods, mainly mahogany (species of the genus *Swietenia* of the Meliaceae family), already started in the 19th century. However, the first forestry development program was not initiated until 1974, with the establishment of COHDEFOR (Corporación Hondureña de Desarrollo Forestal). COHDEFOR quickly became corrupt, and it has been regarded inefficient in controlling illegal logging and deforestation (Merrill, 1995). Similarly to the other countries of the region, Honduras revised its forest laws and policies at the beginning of the 1990s to better incorporate the general guidelines of SFM (ITTO, 2006).

Although the majority of the Honduran forest area is state-owned (FAO, 2006), the rural population's rights to forest resources have been increasingly recognized. Community operators may be given usufruct forest management rights, which necessitate the formulation of a 5-year management plan. In the broadleaved forests, the annual allowed amount of extracted wood is 200 m^3 per forestry group. This amount can in some cases be exceeded, but not over the limit of the sustainable cut defined in the management plan. Harvesting is carried out in segments of 10–20 ha, and the rotation period is commonly set to 30 years. The harvesting system can be perceived to have a relatively low impact on the forest ecosystems. The felling of the trees is commonly done by chainsaws, and they are sawed into cants at the logging site. Mules and rivers are used to transport the cants to the nearest road; where access is difficult, cants are carried on the shoulder.

3.2 STUDY AREAS

The study was conducted in the two areas where FSC forest management certifications had been given to community-based operations in Honduran broadleaved forests (Figure 1). These areas differ from each other in terms of their accessibility and history of forest use, as well as in terms of the main timber tree species that are the focus of logging operations.

3.2.1 RÍO CANGREJAL

The first study area consisted of nine forests (I, II, IV; Table 2) located in the Río Cangrejal watershed area (Figure 1). The mean annual temperature in the area is 26.6 °C (NCDC, 2006; data from 1994–2005), and the average rainfall reaches 2970 mm per year (Vose et al., 1992; data from 1951–1990). The soils are mainly neutral to acidic ultisols with relatively low natural fertility (NRCS, 2005). The forests are classified as tropical moist and premontane wet forests (Holdridge, 1967). Typical forest species include *Euterpe precatoria* Mart., *Vochysia* sp. Aubl, *Genipa Americana* L. and *Terminalia amazonia* (J. F. Gmel.) Exell (Ferrando, 1998).

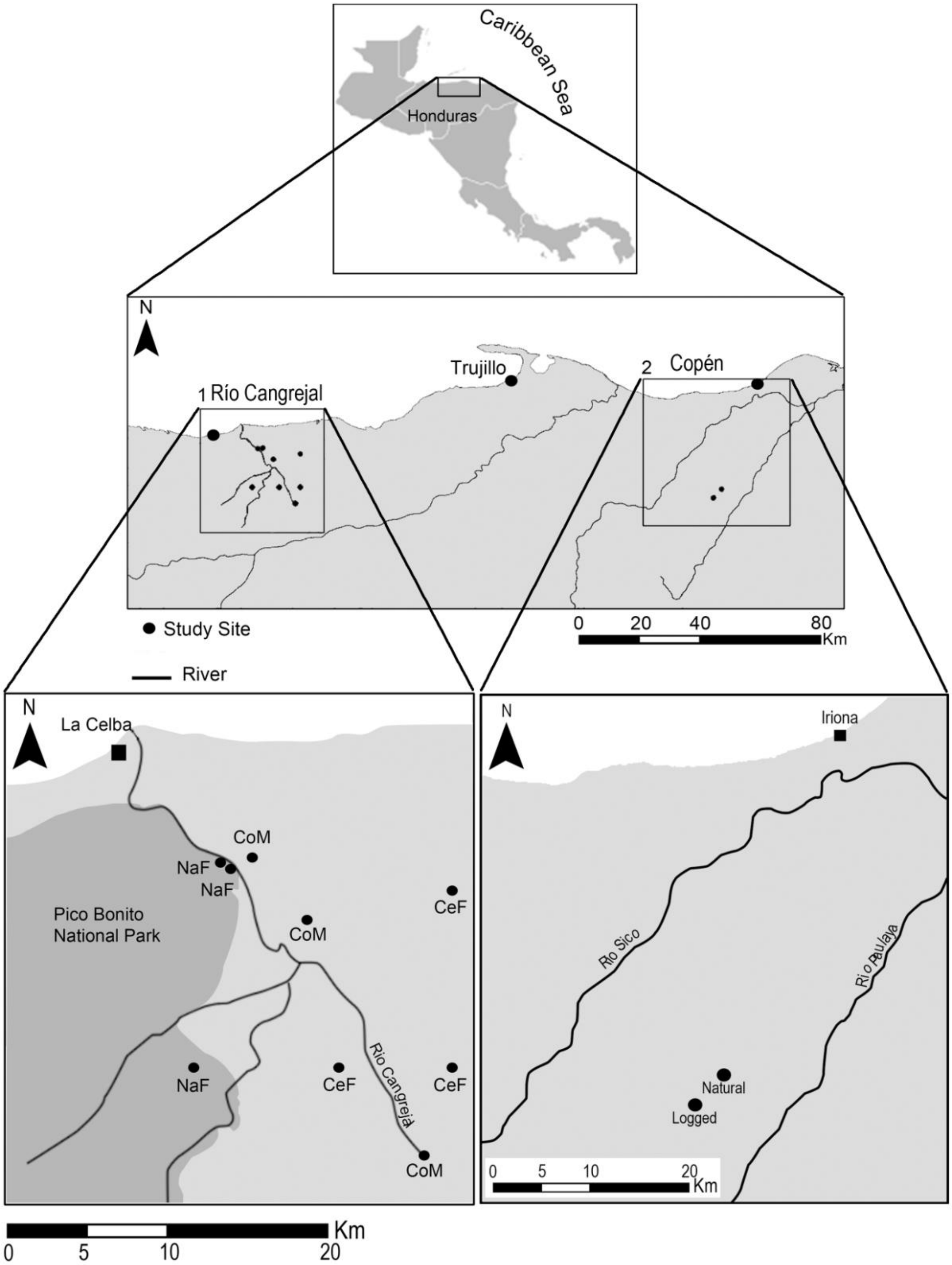

Figure 1. Map showing the location of the two study areas in Honduras (top), in northern Honduras (middle), and the location of the sampled forests in the two study areas (bottom).

The local forestry groups started commercial timber exploitation in the area in the early 1970s. Initially, harvesting focused on the few most valuable hardwoods, *Swietenia macrophylla* King, *Cedrela odorata* L. and *Magnolia yoroconte* Dandy (Markopoulos, 1999). Uncontrolled loggings have since then diminished the resources of these valuable traditional timber species, and current loggings have expanded to cover a range of about 20 species. While most of the lowland areas in Río Cangrejal have been converted to agriculture, forests mainly remain on the steep hillsides.

The three certified community operations included in the study (Table 2) were given their first certificate of good forest management in 1991 through the SmartWood Programme of the Rainforest Alliance under the umbrella organization, Cooperativa Regional Agro-Forestal Colón, Atlántida, Honduras, Ltda. (COATLAHL). Later, these groups were re-certified in 1993, and after SmartWood was accredited by the FSC, they have been re-certified again various times. The forests in Río Cangrejal were considered to fulfil many of the ecological requirements for certified forestry, largely due to the low-impact logging system, and the fact that Honduran forestry law shares many of the certification requirements.

The ecological preconditions set by the SmartWood auditors included improving the planning of seed tree retention and minimum logging diameters to guarantee the successful regeneration of the timber species. To reduce the overall negative effect of human disturbance on the regeneration of the valuable timber species, less time spent on the logging in each forest segment was required. The preconditions regarding technical improvements were related to reducing the disturbance caused by the felling and cutting of timber with chainsaws. Directional felling was required to reduce damage to advance regeneration and to avoid the contamination of water with logging waste. In addition, the focus was on controlling the use of non-commercial tree species and on the conservation of threatened forest species (SmartWood, 1998; SmartWood, 2003). Although all forestry groups in Río Cangrejal fulfilled the requirement of more than 10% of their forests dedicated for protection, the audit team recorded problems with the marking of these areas. In addition, the lack of specific plans for the conservation of threatened species was considered a problem. There were no such plans for the protection of forest fauna from the use of rural populations, either (SmartWood, 1998).

Table 2. Studied forests in the Río Cangrejal and Copén study areas.

Forest type	Study area	Name of forest	Geographic coordinates (decimal degrees)	Size of production forest (ha)	Altitude (m, above sea level)
Certified (CeF)	Río Cangrejal (I, II, IV)	Río Viejo	15.60111, 86.66722	618	930
		Toncontín	15.60083, 86.60111	1061	900
		Yaruca	15.70028, 86.60111	625	650
	Copén (III)	Sanguijuelosa	15.56197, 85.32205	856	325
Conventionally managed (CoM)	Río Cangrejal	El Naranjo	15.71806, 86.71722	1682	250
		El Pital	15.68417, 86.68472	N/A	500
		El Urraco	15.55139, 86.61722	1709	950
Natural/ unlogged (NaF)	Río Cangrejal	Las Mangas	15.71694, 86.71806	-	850
		La Primavera	15.60111, 86.75083	-	200
		Pico Bonito	15.71722, 86.73389	-	200
	Copén	Marañones	15.59180, 85.30014	-	330

3.2.2 Copén

The second study area comprises two forests (III, Table 2) that are located in the Sico-Paulaya valley in north-eastern Honduras (Figure 1). The mean annual temperature is 26.6 ºC, and the rainfall varies between 2850–4000 mm per year (Herrera-MacBryde, 2004). The studied forests belong to tropical moist forests (Holdridge, 1967), and typical species include *Albizia carbonaria* Britton, *Calophyllum brasiliense* var. *rekoi* (Standl.) Standl., *Cecropia* spp. Loefl., *Ficus* spp. L., *Luehea seemannii* Triana & Planch., *Inga* spp. Mill, *Lonchocarpus* spp. Kunth, *Ochroma lagopus* Sw., *Pachira aquatica* Aubl., and *Heliconia* spp. L. (Herrera-MacBryde, 2004).

Unlike Río Cangrejal, this area is located in the frontier of agricultural conversion of primary forest areas. The studied forests belong to the buffer zone of the Río Plátano

Biosphere Reserve (RPBR), which was declared a UNESCO Biosphere Reserve in 1980. The RPBR covers approximately 500 000 hectares of land, being the largest undisturbed forest area in Central America (UNEP-WCMC, 2009). Although the area is relatively sparsely populated, settlements of migrant farmers are spread across the buffer zone, extending to the forest reserve. Secondary forests are abundant in the inhabited areas and abandoned agricultural land, typically hosting species such as *Salix humboldtiana* Willd., *Ceiba pentandra* (L.) Gaertn., *Bambusa* spp. Schreb, and *Pithecellobium* spp. Mart. (Herrera-MacBryde, 2004). Due to the lack of major roads, the Sico-Paulaya valley hosts relatively dense populations of *S. macrophylla*. Illegal logging is, however, prevalent in the area. The problems for local communities are intensified by the linkages between illegal logging and other illegal activities, such as robbery and drug-trafficking, causing outbursts of violence (Del Gatto, 2004).

The study areas are managed by the community forestry group of Copén. The group completed its first management plan in 1998 (Pavón and Peralta, 2004). In the same year it was granted the FSC forest management certificate through the SmartWood Program of Rainforest Alliance. The certification has, however, been discontinued due to the lack of financial resources and organisational constraints. The main difference in the logging system, as compared to that of Río Cangrejal, is that the logging focuses on a single species. *Swietenia macrophylla* is used for various purposes within the village, as well as being logged for commercial marketing. However, in the management plans, quotas are also set for the cutting of other marketable and potentially marketable timber species (Peralta, 2005).

Similarly to Río Cangrejal, the logging in Copén may be considered to have a relatively low environmental impact. Pre-logging inventories are conducted, and in Sanguijuelosa, the minimum logging diameter of *S. macrophylla* is set to 80 cm (Peralta, 2006), which is high compared to most other areas where the species is logged. In the biannual operative plan of 2006–2008, 65% of the mature *S. macrophylla* individuals were selected as retention seed trees (Peralta, 2006). Mechanical logging damage is controlled by practicing pre-logging cutting of lianas (Pavón and Peralta, 2004), restricting tree felling on steep slopes, and minimizing path construction (Peralta, 2005).

In the certification assessments (SmartWood, 2002), deficiencies were found in the monitoring of forest regeneration and growth rates. The logging of only one species, *S. macrophylla*, was considered a problem, and the need to expand the range of logged species was emphasized. Requirements were placed to improve the planning of directional felling to minimize damage caused to residual trees. Furthermore, specific attention was paid to the need for guarding the forests against illegal logging and hunting. Several requirements were made for improving the status of biodiversity protection; inventories on the distribution of vulnerable fauna were required, as well as a plan of action for protecting these species. Furthermore, deficiencies were found in planning the protection of vulnerable sites, such as riverbanks (SmartWood, 2002).

3.3 STUDIED SPECIES

A full list of the studied species is provided in the Appendix. The species were classified as timber or non-timber species, and when possible, according to their light ecology (see Chapter 1.2.2) (Appendix).

3.4 SAMPLING METHODS

3.4.1 RÍO CANGREJAL

In Río Cangrejal, the studied forests included three CeFs, three CoMs and three NaFs (Table 2, I, II, IV). In each CeF and CoM, five logging gaps were sampled; correspondingly, five to six natural treefall gaps were sampled in each of the NaF sites. The gaps were selected so that they represented single treefalls (although in some cases there were few collateral treefalls), and were not situated near forest edges or openings. In each gap, three round 100-m^2 plots were drawn, situated in the stump, bole and crown sites of the fallen tree. This type of sample taking was performed because the different gap zones typically exhibit floristic differences (Brandani et al., 1988).

In each sample plot, all woody plants were recorded according to their local names (see Appendix). The size of the recorded plants was limited to ≥ 1 cm at dbh, because this is commonly considered the limit after which the probability of sapling survival increases considerably (Delissio et al., 2002; Obiri and Lawes, 2004). Samples of the recorded species were taken to the Paul C. Standley Herbarium (EAP) in Zamorano, Honduras, where 54 taxa (52.7% of those identified with local names) were taxonomically identified to species or genus level.

A set of environmental variables was additionally recorded in the sampled treefall gaps. Gap age was estimated with the help of local forest group members. Gap age affects the species composition within the gaps. Immediately after gap formation, pioneer tree species dominate gap regeneration, whereas during later years, the relative proportion of shade-tolerant tree species increases (Brokaw and Scheiner, 1989; Dupuy and Chazdon, 2006). Gap size in turn affects the amount of incoming light and thus the vegetation within a gap, particularly the relative proportions of light-demanding and shade-tolerant species (e.g. Barton et al., 1989). Due to having differently aged gaps that were in different stages of canopy closure, gap size was measured based on the estimated size of the fallen tree to obtain an estimate of the light conditions at the time of gap creation. Altitude (above sea level) was recorded, as it is directly related to temperature and rainfall patterns, and thus affects the distribution of many plants. Inclination also affects the distribution of many plant species (Lieberman et al., 1996; Wadsworth, 1997).

The percent coverage of stones, dead wood and litter were measured, as these affect the available growth space. The total stem density and percent coverage of lianas, herbs, shrubs, and trees were also measured, because these may compete with the regenerating timber species. As an indicator of conditions favourable for the pioneer species, the abundance of *Heliocarpus appendiculatus* Turcz was recorded. The locations of the studied forests were recorded with a global positioning system (GPS) receiver. Increasing distance can be expected to affect the floristic composition in gaps due to 1) biotic factors related to seed dispersal barriers and dispersal distances and 2) abiotic environmental variation such as substrate type (e.g. Condit et al., 2002; Slik et al., 2003).

3.4.2 COPÉN

In Copén, the studied forests included a CeF and a NaF protected from logging (Table 2, III). Fifteen logging gaps were sampled in the CeF and 10 natural treefall gaps in the NaF. Similarly to the sample collection in Río Cangrejal, three round sample sites were situated in the stem, bole and crown of the fallen tree. However, due to the rather small size of the area disturbed by logging, and the dense layer of vegetation, the sample sites were limited at 10 m^2 in size. Again, all woody species were recorded and identified to species with local names. Scientific identification based on collected specimens was carried out at the ESNACIFOR herbarium. The recorded individuals were divided into two size groups: seedlings (20–150 cm in height), and saplings (>150 cm). The recorded environmental variables included gap age, inclination, canopy openness at gap and subgap levels, the dominant vegetation layer, density of seedlings and saplings, presence of an abundant shade-tolerant palm, abundance of lianas, and the percent coverage of stones, dead wood, bare soil and litter (I, II, III).

3.4.3 LIMITATIONS OF THE SAMPLING METHODS

Conducting a study on the ecological impacts of ‘on-the-ground’ forestry practices proved challenging in many ways. The study area of Copén was difficult to access, and due to prevalent illegal logging activities, only the forests managed by one community could be studied, and no replicate forests were sampled as in Río Cangrejal. Due to the short history of loggings in Copén, the sample size remained limited, and consequently, the results have to be interpreted with caution. In Río Cangrejal, a larger set of forests could be sampled (Figure 1), although as many of the shade-tolerant timber species were relatively rare, the sample size again remained relatively small.

Due to the limited amount of suitable CeFs, CoMs and NaFs in the studied areas, it was impossible to find sample sites that would have been similar in regard to the environmental conditions that may impact species composition within canopy gaps. For example, the forests were located at different altitudes, and the mean values of for example gap age, inclination and stone coverage varied between the CeFs, CoMs and NaFs (I, II, III). Their impact could have been erroneously interpreted to have been caused by the differing management type. To avoid this kind of misinterpretation of the results, the impact of these covariates was factored out in the statistical analyses, before analysing the impact of the variables related to the management regime (see Chapter 3.5). However, some environmental gradients that may act as important determinants of tree species composition, such as soil fertility, drainage and texture (Clark et al., 1999; de Carvalho et al., 2000; John et al., 2007), were not measured in this study.

The logged forest area in Copén was located at a considerable distance from the protected site (Figure 1). Areas further apart are more likely to differ in environmental conditions and species composition, and these differences could erroneously be interpreted as being caused by a different management regime. In Río Cangrejal, the CeFs were generally located further from the Pico Bonito National Park than the CoMs, of which only one was not located in close proximity of the park. This means that the CoMs may be more influenced by seed dispersal from the NaFs (II). Liu and Ashton (1999) found that seed dispersal from surrounding species-rich areas may significantly compensate the homogenizing effect of logging on the floristic composition. The proximity of NaFs may be particularly important in regard to the heavier, gravity or animal-dispersed species, which often have clumped distributions and land close to the mother tree (Harms et al., 2001). Many seed dispersing animals may also avoid crossing open areas (Holl, 1999). Species with smaller, wind-dispersed seeds may be distributed more randomly (Dalling et al., 2002). To avoid misinterpretation of the results due to the spatial proximity, distance was added to the analysis as a covariate to account for its effect (II). However, it must be noted that the distance variable only covered the physical distance between the studied forests (II), while a measure capturing their connectivity could have been ecologically more relevant.

The lack of follow-up data can also be considered a limitation in this study. The studied gaps were in the early stages of succession, which means rapid changes in the community composition. Generally, the first years after gap creation are dominated by pioneer vegetation, after which, the conditions become increasingly shaded. This causes many of the light-demanding species to die off, and the shade-tolerant species start becoming relatively more abundant (Brokaw and Scheiner, 1989). It proved impossible to find enough gaps of the same age in the Río Cangrejal forests, and hence, we attempted to account for the differences in successional stage by including the covariate gap age in the analyses (I-III). Furthermore, the light-demanding species were excluded from the analyses of timber species regeneration success (I), since their abundances are probably more affected by the successional stage of the gap (Brokaw and Scheiner, 1989).

The studied forests were poorly known to such an extent that very little basic scientific data were available. The past use histories and other factors that may strongly affect vegetation patterns could only be speculated. This problem is, obviously, not unique to this study – almost all forests are to some extent affected by past human disturbance, and distinguishing this impact from past natural disturbances may be impossible (e.g. Chazdon, 2003). Furthermore, although the protected forest areas were assumed in this study not to be affected by human use, illegal logging may in fact be relatively common within the protected areas in Honduras. The Marañones site (III) was a recently designated protected area that was situated close to the agricultural frontier and had thus probably experienced extraction of valuable hardwoods.

3.5 STATISTICAL ANALYSES

To test the hypotheses on regeneration success (RSa, RSb and RSc), Poisson regression analyses were conducted (I, III). The counts of seedlings and saplings of timber species in the studied subgaps were handled as repeated measures. The data from both study areas were found to be 'zero-inflated' i.e. the proportion of zero values was higher than assumed for a Poisson distribution. Zero-inflation is often found in ecological data containing species that occur in low frequencies (Dobbie and Welsh 2001; Fletcher et al. 2005; Martin et al. 2005). Statistical analyses that allow for zero inflation have been developed; however, here, a more ecologically sound approach was chosen. It was assumed that the mechanisms behind the occurrence and abundance of saplings are different (Ridout et al. 1998; Cunningham and Lindenmayer 2005). Whereas the occurrence of saplings in the treefall gaps in tropical forests may be expected to be more dependent on the abundance and distribution of seed trees in the forest, their abundance may be more dependent on the characteristics of the habitat, such as the availability of light and other resources. Hence, the data were divided into two sets: logistic regression was used for the occurrence data (presence or absence of species) and Poisson regression for the abundance data (total count, excluding zero observations) (I, III).

Furthermore, problems of 'overdispersion' arose in the data, typically caused by correlation between observations (McCullagh and Nelder, 1989). Due to e.g. the distribution of seed trees and soil characteristics, one could expect the observations within a gap or within a forest area to be correlated. To overcome problems related to overdispersion, generalised estimating equations (GEEs) were used (I, III) (Liang and Zeger, 1986). Furthermore, mixed models were used with gap as the random effect to study the importance of the 'unknown effect' i.e. gap characteristics that were not included in the measured environmental variables (Lee and Nelder, 2000).

To test the FS hypothesis, ecological ordination was used (II). The linear form of constrained ordination, redundancy analysis (RDA), was selected to compare the floristic similarities between the forest management types (CeF–NaF and CoM–NaF). Principal components analysis (PCA) was used to test the effect of management type in an unconstrained ordination space, i.e. to allow for that part of the variation that was not explained by the management type (II). PCA and RDA allow the presentation of sample sites in two-dimensional graphs where sites with the smallest differences in species composition are located closest together and sites with the most dissimilar composition are located furthest from each other.

Both PCA and RDA display the Euclidean distance among sites. It has been pointed out that the Euclidean distance has limitations when comparing species abundances, especially when the data contains null abundances; sites sharing no common species can in some cases be interpreted as being more similar to each other than sites sharing species (Legendre and Legendre, 1998). However, PCA and RDA have been considered as suitable methods for data with relatively low beta diversity, i.e. low species turnover (ter Braak and Šmilauer, 2002). According to ter Braak and Šmilauer (2002), the suitability of a linear ordination method can be tested using Correspondence Analysis; here, DDCA (Detrended Correspondence Analysis) supported the selection of a linear ordination method (gradient length was < 3, reflecting a relatively low beta diversity)

To test the hypotheses on gap favourability (GFa, GFb, GFc and GFd) (I–III), the environmental gap characteristics were divided into two subgroups. This division was based on the expected impact of forest management. The management-sensitive (MS) gap characteristics group included those variables that could be expected to be affected by the forest management type. Correspondingly, the management-neutral (MN) gap characteristics included those variables that could not be expected to be affected by the forest management type. To eliminate initial differences in the MN gap characteristics, these were added to the regression models first as explanatory variables. After this, the MS gap characteristics were added as explanatory variables to examine the extent to which they explained the variation in regeneration between forest management types (I, III). Correspondingly, partial redundancy analysis (pRDA) was used to extract the variation explained by the MN and MS gap characteristics from the floristic data matrix before studying the residual variation with respect to the variables of interest (II).

4. MAIN RESULTS AND DISCUSSION

4.1 REGENERATION SUCCESS OF TIMBER SPECIES

4.1.1 SHADE-TOLERANT TIMBER SPECIES

The results obtained in Río Cangrejal did not support hypothesis RS(a), as there were fewer shade-tolerant timber tree species and individuals in CeF than in CoM (Table 3) (I). This result suggests that certification has been an inefficient tool in guaranteeing the sustained regeneration of the timber tree species, and moreover, that conventional management would be a better means of enhancing the regeneration of the valuable species. However, the results were in concordance with hypothesis GF(a); when adjusting for the MS covariates in the regression models, the difference in regeneration success between the CeFs and CoMs increased (Table 3) (I). This indicates that the CeF gaps created a more favourable environment for the regeneration of shade-tolerant species than the CoM gaps.

In the light of the assumptions based on earlier studies, many of the MS characteristics of the CeF gaps in Río Cangrejal indicated reduced logging damage compared to the CoM gaps; on average, the CeF gaps had a higher coverage of woody stems and trees, lower liana and shrub coverage, and hosted a lower abundance of the pioneer species *H. appendiculatus* (I). However, this observed lower damage should be interpreted with some caution. First, the differences in tree and liana coverage between the CeFs and CoMs were negligible, which may be explained by the frequent pre-logging removal of lianas, practiced in both CeFs and CoMs (I). Second, the *H. appendiculatus* and shrub abundances were not significantly associated with the regeneration success of the shade-tolerant timbers. Third, contrary to expectations, herb coverage was positively correlated with the abundance of shade-tolerant species, and higher in the CeFs than CoMs (I). Fourth, although increasing stem density was positively associated with regeneration success, this variable turned out to be an ill-suited measure of the post-logging survival of advance regeneration. The increase was largely due to few particularly abundant pioneer species, such as *Cephaelis* sp., *C. tomentosa* (II), *Calathea sp.* and *P. aduncum* (III). This suggests that a major part of the stems were generated after gap formation, rather than representing advance regeneration.

Furthermore, the observed differences in the regeneration success of the shade-tolerant species between the CeFs and CoMs could to a large extent be explained by one of the studied shade-tolerant species, *M. vestitum*, being particularly abundant in the CoMs. This species has a poorly known light ecology and it may possess many characteristics of the light-demanding species, hence possibly benefiting from a higher level of logging damage in the CoMs.

To explain the remaining differences in the regeneration success of the shade-tolerant species, it was suggested that the pre-certification loggings in the CeFs had been more intensive or started earlier than the loggings in the CoMs (I). Since good community organisation is required of the certified communities, the CeFs may also have a longer or more intensive history of pre-certification forest exploitation than the CoMs (II). As a result, the regeneration of the shade-tolerant species would be limited by the impacts of past logging on the abundance of reproductive individuals. This could explain why, although the current conditions in the CeF gaps indicated reduced logging damage, the regeneration success of the shade-tolerant timber species was still lower in the CeFs than the CoMs (I). Furthermore, in ecological terms, certification was initiated relatively recently in the study areas, and during the first years the implementation of the certification criteria was less strict (Markopoulos, 1999). The impacts of RIL and reduced harvest intensities would not yet be likely to show in the mature tree species composition.

The observed higher regeneration success of the shade-tolerant species in NaFs compared to CeFs and CoMs indicates that logging had reduced the populations of the shade-tolerant species in both managed forest types (Table 3) (I). The results in Río Cangrejal (I), as well as in Copén (III), support hypothesis RS(b). However, hypothesis GF(b) was not supported by the results in either study area; the NaF gaps did not provide more favourable environments for the regeneration success of the shade-tolerant species than the CeF gaps. This result may partly reflect the lack of unambiguous measures of disturbance, as discussed earlier in this chapter.

4.1.2. LIGHT-DEMANDING TIMBER TREE SPECIES

Hypothesis RS(c) was partially supported by the results; in Copén, there were more species and individuals of light-demanding timber tree species in the CeF than in the NaF gaps, with a statistically significant difference in abundance (Table 3) (III). However, the main commercially valuable species, *S. macrophylla*, was present in only one of the NaF gaps, whereas in the CeF, it was found in over half of the gaps. Furthermore, there was reason to doubt whether the regeneration of *S. macrophylla* was successful in the CeF gaps, regardless of its relatively high abundance. In the established (4-year-old) logging gaps, *S. macrophylla* seedlings were overtopped by a dense vegetation layer closing at a height of 3–4 m (III). Based on earlier studies, *S. macrophylla* seedlings deprived of light have little chance of maturation (Grogan et al., 2005; Snook and Negreros-Castillo, 2004).

Hypothesis GF(c) was also partially supported by the results. Accounting for the effect of the MS gap characteristics explained only 15% of the difference in light-demanding species occurrence, but the difference in abundance between the CeFs and NaFs was fully explained (III) (Table 3). As the NaF was located closer to the expanding agricultural frontier than the CeF, it seems likely that those differences in light-demanding timber tree regeneration success between the CeF and NaF not explained by the GF could be due to pre-certification loggings in the NaFs.

Table 3. A summary of the results of hypothesis testing. Single degree-of-freedom contrasts were used for the species occurrence and sapling abundance models. For hypothesis RS(a), CeF (certified forest) and CoM (conventionally managed forest) were contrasted (with contrast coefficients 1 and -1), and for hypotheses RS(b) and RS(c), CeF and NaF (natural forest) (coefficients -1 and 1; 1 and -1, respectively). The contrasts were tested against zero in three cases: with no gap characteristics as covariates, using only MN (management-neutral) covariates and finally also adding MS (management-sensitive) covariates to test hypotheses GF(a), GF(b) and GF(c). Estimates in the occurrence (logit) model are depicted as odds ratios, and in the abundance (Poisson) model as the ratio of the mean number of saplings (e.g. the predicted count for CeF saplings with all covariates included is 0.52 times that of CoM). P-values are based on Wald chi-square statistic; those below 0.05 are given in bold. Odds ratios below 1.0 are represented using their inverse values. This makes their comparison with odds ratios above 1.0 easier (Rita and Komonen, 2008).

Hyp.	Contrast	Covariates	Occurrence		Abundance	
			Estimate	P-value	Estimate	P-value
RS(a)	CeF vs. CoM	MN	1.30^{-1}	0.2730	0.54	**0.0035**
GF(a)		MN+MS	1.99^{-1}	**0.0166**	0.52	**0.0073**
RS(b)	CeF vs. NaF	MN	2.48	**0.0091**	1.87	**0.0496**
GF(b)		MN+MS	3.08	**0.0003**	1.45	0.1122
RS(c)		MN	4.48	**0.008**	1.14	0.70
GF(c)		MN+MS	3.83	0.06	0.30	**0.002**

4.2. The naturalness of the floristic composition

The CoM gaps were floristically more similar to the NaF gaps (Figure 2a), whereas the CeF gaps hosted the highest proportion of the most common light-demanding species. Furthermore, the gap conditions in the CeFs were less favourable for a natural-like floristic composition than the gap conditions in the CoMs (Figure 2b) (II). These results were in contrast with hypotheses FS and GF(d), suggesting that the CeF management practices do not support the aim of maintaining a NaF species composition.

However, as the CeF gaps showed reduced logging disturbance (I), and many significant factors potentially affecting the floristic composition were excluded from the analyses (II), alternative explanations need to be discussed. As suggested in Chapter 4.1, pre-certification loggings in the CeFs of Río Cangrejal could explain the higher abundance of many non-commercial light-demanding species (see I). Furthermore, the post-disturbance recovery of NaF species composition is dependent on the proximity of undisturbed forest areas (Chazdon, 2003); hence, the CeFs, being located further away from the NaFs, were probably relatively more influenced by seed dispersal from agropastoral areas than the CoMs, which were generally located closer to the NaFs. The high abundance of light-demanding, early-successional species that are typically found in agropastoral areas in the CeF gaps provide further support for this assumption (II).

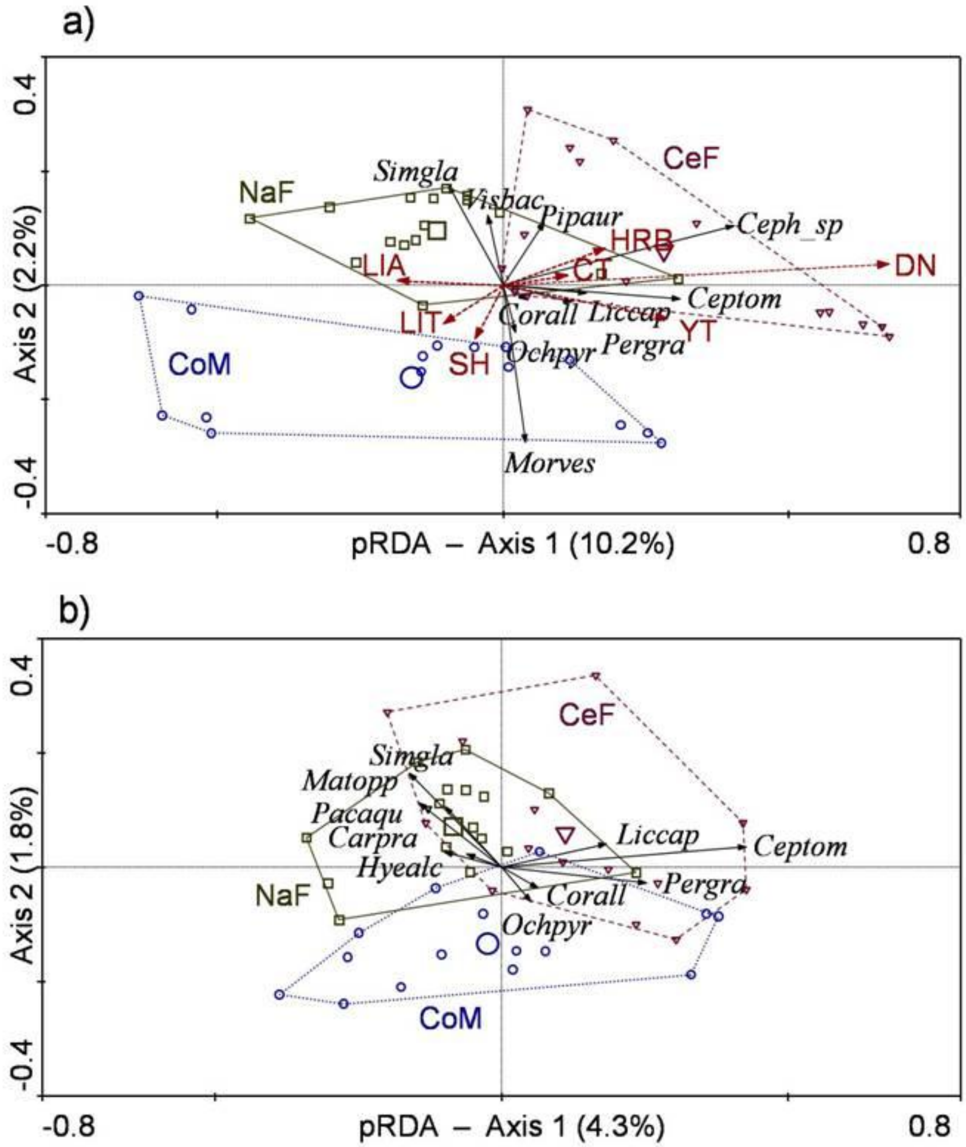

Figure 2 a and b (II). Partial redundancy analysis (pRDA) ordination diagram, plotting sample (gap) scores (small symbols) and class centroids of forest management type (large symbols) along with 10 taxa with the best fit. Envelopes are drawn around samples of each management type, with sample scores represented as symbols (CeF = down triangle, CoM = circle, NaF = square). Axes 1 and 2 are constrained, with management type used as the explanatory variable. The percentage of variance in the floristic dataset explained by each axis is indicated in brackets. Management-neutral (MN) gap characteristics are used as covariables. Management-sensitive (MS) gap characteristics are used as supplementary variables, added to the ordination *post hoc*. Abbreviations for MS gap characteristics are as follows: DN = density, HRB = herb coverage, SH = shrub coverage, LIT = litter coverage, LIA = liana coverage, CT = coverage of canopy trees, YT = coverage of young trees.

4.3. Practical implications: the feasibility of FSC certification criteria in community operations

The results of the ecological study indicate that certification has not, at least yet, succeeded in significantly improving the ecological sustainability in the studied systems (I-III). Based on the results, it was further hypothesized that the past forest uses and the fragmentation of the remaining forest areas may constrain post-logging forest recovery in the studied areas (I-II), and that certified logging may create environmental conditions that do not support the regeneration of some economically valuable timber species (III).

At the same time, it is important to note that certification has brought many benefits to the communities in the Río Cangrejal study area. The recognition of customary resource rights has improved, and the governmental approval of the usufruct contacts has become easier. The forestry groups' participation in forest decision-making has increased, and FSC certification has served as documented evidence of their good management practices (IV). The increased awareness among the community members of the value of their forests, and the improved control over unauthorized forest clearing and illegal logging are factors that can also be expected to have positive impacts on the ecological sustainability of forestry operations.

The recommendations to improve the certification of community-based forest management systems in the tropical areas were summarized as four main issues that need increased attention, namely: 1) the high level of heterogeneity in local forest ecosystems and communities; 2) the history of forest use; 3) forest ecosystems and forest activities as part of the eco-social landscapes, and 4) links to the wider political economy. Each of these factors reveals important linkages between the ecological and the socio-economic issues.

4.3.1 High level of heterogeneity in local forest ecosystems and communities

The results (I, III) indicate that certified forest management may enhance the regeneration of certain tree species, whereas at the same time limiting the regeneration of others. This is due to the variation between species in the response to logging disturbance, and it makes it difficult to reliably predict the impacts of certified management on guaranteeing a sustainable yield of timber. Furthermore, some timber tree species may be particularly difficult to log in a sustained manner (III). These are often species that have a regeneration strategy that makes them reliant on disturbances larger than those that are created by RIL. Therefore, these species have been suggested to need silvicultural interventions to regenerate successfully (d'Oliveira, 2000; Grogan et al, 2002; Grogan and Landis, 2009; Snook and Negreros-Castillo, 2004; van Rheenen et al., 2004) (III, IV). However, such interventions may lead to intensified competition from fast-growing secondary species, especially in fragmented forests, which are vulnerable to the colonization of species from the surrounding agropastoral areas (II). Similarly to Río Cangrejal, many community forest landscapes in the tropics consist of forest patches of differing sizes in a matrix of agropastoral land. As the conversion of forests to agropastoral land continues, such areas will become increasingly common.

Similarly to the ecological systems, heterogeneity is characteristic of the social systems of community-based forestry. Forestry is usually one amongst a variety of economic activities and income sources, and its role in income generation may be highly sporadic, although often very important. The various resource users have different interests and priorities in terms of forest use, which limits the application of criteria that treat communities as a homogeneous group (IV).

In Río Cangrejal, as well as in Copén, little scientific knowledge exists on forest composition and structure. Even so, FSC criteria require evaluation of the impacts of logging on forest structure, functions and biodiversity at all levels. This is a particularly demanding task in small communities in the tropics, which suffer from a lack of resources and formal scientific expertise to conduct such evaluations in the highly diverse ecosystems.

4.3.2 THE HISTORY OF FOREST USE

The impacts of logging are not only dependent on the harvesting methods, but are also significantly affected by the pre-harvest conditions in the forests (Guariguata and Pinard, 1998). Our findings indicate that selective logging in previous decades may have negatively affected the populations of many non-traditional (I), as well as traditional timber species (Markopoulos, 1999) (III). The problem of managing the forest for species with different responses to logging (Chapter 4.3.1) is particularly difficult in areas such as Río Cangrejal, where the exhaustion of the few most valuable timber tree species has forced the communities to expand loggings to cover a range of non-traditional species. This also limits the economic profits gained from forestry, as the non-traditional species have a limited market and generally lower price. Using ecological information as a basis for identifying and conserving those species that are most vulnerable to the impacts of logging (as suggested by Jennings et al., 2001) is unlikely to succeed as long as the market demand remains unaffected.

A similar situation may be found in many other tropical community forests, because of intensive periods of selective logging and third-party illegal forest exploitation at times when local resource rights were poorly legitimated (Taylor, 2005b). In such cases, RIL-type requirements of a reduced mechanical impact and improved harvest planning emphasized in the certification guidelines may not be sufficient in restoring the timber tree populations, but active restoration efforts may be required (IV).

On the other hand, the human impact on tropical forest ecosystems has in many cases continued for centuries; systems of small-scale timber production, agriculture and NTFP gathering within the forested areas may maintain high landscape-level biodiversity. Furthermore, local communities may have significant traditional ecological knowledge that would potentially benefit the planning of sustainable forest management, if better incorporated in certification standards (IV).

4.3.3 FOREST ECOSYSTEMS AND FOREST ACTIVITIES AS PART OF ECO-SOCIAL LANDSCAPES

As discussed in Chapters 4.3.1–4.3.2, certified community-forest landscapes can display a degree of degradation and fragmentation due to past resource uses. In such areas, many of the processes affecting the ecological sustainability of forestry act beyond the forest management unit (FMU) level, on which the certification standards are applied (IV). The effects of forest fragmentation may be particularly detrimental to the tree species of humid tropical forests, due to high levels of between-population gene flow required to maintain population viability (Alvarez-Buylla et al., 1996). Furthermore, the shade-tolerant species, found to have a low regeneration success in the CeFs, typically have larger seeds and poorer dispersal abilities than many light-demanding species (Canham, 1989).

In fragmented forest areas, the post-logging recovery of the forest is affected by the quality of agropastoral areas as migration pathways for forest organisms (IV). Many practical actions may be taken in agropastoral areas to improve the sustainability of forest management. Suggested improvements include the establishment of small-scale agroforestry systems where timber trees are planted on agropastoral lands (Harvey et al., 2008; Martínez-Garza and Howe, 2003), silvopastoral systems where timber is produced on pasture (Perfecto and Vandermeer, 2008), and the maintenance of gallery forests along waterways on farmlands (Tabarelli and Gascon, 2005). In the Copén study area, the abundance of *S. macrophylla* was seemingly higher in the secondary than the primary forest areas (III). Similar observations concerning African mahogany species led Makana and Thomas (2006) to suggest that secondary forests may have a potentially high significance in timber production and forest conservation.

The focus of certification on the FMU level also limits the opportunities for efficient biodiversity conservation. Although the requirements for set-asides within the certified forests may enhance the survival of many primary forest species, the use of certified forests as buffer zones of protected areas might potentially have a more profound impact on the conservation of biodiversity (Azevedo-Ramos et al., 2006; Putz et al., 2001). It has been shown that the intensified land use around protected areas in the tropical countries may compromise the effectiveness of these areas in biodiversity conservation (Ewers and Rodrigues, 2008). This 'leakage' effect could potentially be controlled through supporting forest certification in the immediate surroundings of protected forest areas. This assumption, however, relies on forest certification being an efficient tool in preventing deforestation. The interviews conducted in the communities of Río Cangrejal indicated that at least in this case, certification had reduced illegal conversion of forests to agropastoral areas (IV).

Similarly to the ecological processes, the socio-economic processes related to sustainability reach beyond the isolated view of communities reflected in the current FSC certification standards. Whereas the standards set requirements for the community forestry groups to document their resource rights, little has been done to increase the governmental authorities' responsibilities to ensure the security of resource rights. In community operations, a particularly serious problem is that certification directs the responsibility for controlling illegal harvesting to communities (IV).

4.3.4 Links to the wider political economy

The specific needs of community forest producers may be poorly recognized in the national forest policies. Hence, the certification requirement of compliance with the national laws may in some cases hinder the opportunities for sustainable forestry in community operations. For example, in the Honduran system, the usufruct contracts are granted to community forestry groups on state-owned lands for a period of only four years at a time, which is too short for the planning of sustainable forest management (IV). A further example shows that in Copén, the strict limits on the amount of wood that the local communities were allowed to remove from the forests led to the piling of sawed planks, often in the middle of the logging gaps; i.e. those sites that are most important for the regeneration success of many light-demanding timber species (III).

Actions to improve sustainability on fragmented and degraded forests include the planting of trees on farmlands and the tending of the regenerating forest trees (Chapter 4.3.3). Such actions are, however, restricted by the lack of political support and economic incentives, and particularly by the insecurity of resource rights in many tropical forest communities (IV). Furthermore, the social benefits gained from certified production are limited by the community producers' poor competitive abilities and lack of market access. Although forest certification undoubtedly has limited possibilities to change the structures of the existing timber market chains, the FSC standards should be better adapted to capture the benefits and constraints faced by Southern community forestry operators in the markets of certified products (IV).

5. CONCLUSIONS AND FUTURE RESEARCH PERSPECTIVES

Whether forest certification will be successful in enhancing the sustainable management of tropical forests depends on its spatial coverage and its ability to affect those processes that are critical to the ecological and social sustainability of forest use. So far, only a fraction of community forests have been certified, although the role of communities as managers of tropical forest resources is rapidly increasing in importance. While the barriers to certification uptake in community forests are relatively well studied, the question of the feasibility of forest certification standards in tropical community-based management systems has so far received little consideration.

The ecological requirements of certified forestry emphasize the use of RIL guidelines to minimize the change in forest structure and species composition. In the studied FSC-certified areas in Honduras, a comparison of the environmental characteristics between certified and conventionally managed forest logging gaps indicated lower mechanical logging damage in the certified forests. Despite this, the regeneration success of the shade-tolerant timber tree species was poorer in the certified forests compared to the natural or conventionally managed forests, whereas the regeneration success of light-demanding timber tree species was higher in certified than natural forests. Furthermore, the floristic composition in the gaps of conventionally managed forests was more natural-like than that in the certified forests.

Two conclusions can be drawn from these results. First, the main constraint to post-logging recovery in certified forests may be linked to forest degradation and fragmentation rather than the impact of current, certified logging. The certified forests in the study area of Río Cangrejal, inhabited for several decades, may have been subjected to more intensive or longer periods of logging than the conventionally managed forests in the past. The difference in logging history may coincide with the strong community organization required from the certified forests. Although these kinds of linkages remain to be proven, it is likely that due to the multiple problems with resource use and land use tenure, much of the area under community forest management has been unsustainably managed in the past. Furthermore, in Río Cangrejal, the certified forests were generally located further away from the protected areas than the conventionally managed forests. The area is characterized by agriculture and cattle rearing, with forests remaining mainly in the steep hillsides. In this landscape, the certified forests may be relatively more affected by seed dispersal from the agropastoral areas, whereas the post-logging recovery of the conventionally managed forest areas may be significantly enhanced by the proximity of pristine forest areas.

Second, the management of light-demanding species in forests with a lesser degree of degradation and fragmentation may be incompatible with the main certification requirements related with the reduction of mechanical logging damage. Although the light-dependent species had a generally higher regeneration success in the certified forests of the Copén study area, the results indicate that the most valuable timber species, *S. macrophylla*, may be unable to regenerate successfully under the certified management regime.

Based on these conclusions, the most serious limitation for the feasibility of FSC certification in community-based systems may be its lack of consideration of forest use history and landscape-level land use patterns. To improve the recovery of degraded forests, actions would need to be taken on the agropastoral areas to improve connectivity between the remaining forest areas. For those species that require large-scale disturbances for successful growth, the maintenance of sustained yield should be balanced against the need to minimize logging-related disturbance. Again, a potential solution could involve linking the agropastoral and secondary forest areas to the management of light-dependent species.

It is with these kinds of recommendations that the linkages to the social aspects of forest management become fundamentally important. The core of FSC forest certification is the general standards that can be applied to any type of forest operations. However, community forestry typically operates in highly complex and heterogeneous conditions: poorly known ecosystems and timber species, a wide variety of stakeholders with different interests towards the forest resource, and political and economic conditions that provide little support for small-scale producers. Any guidelines dealing with the use of agropastoral land areas, for example, are unlikely to succeed without efficient incentives to guarantee the involvement of farmers. Furthermore, a lack of long-term tenure rights for the forests will limit any attempt to restore the degraded forest species. These considerations underline the need for multidisciplinary studies regarding community forestry. Research should also provide practical recommendations, recognizing that the certification standards inevitably have conflicting requirements, such as allowing levels of hunting that meet social requirements while maintaining natural forest biodiversity (Nussbaum and Simula, 2005).

In searching for optimal ways to maximise the biodiversity conservation benefits and social benefits related to certification, a particularly interesting point of further study would be the potential of using certified forests to reduce deforestation and the degradation of forests around conservation areas. Such linkages may also prove to be important in the context of the global carbon-management scheme, where the role of tropical forests (UNECE/FAO, 2008), and more recently, tropical communities as their managers, has been emphasized (Chhatre and Agrawal, 2009).

Further ecological studies that attempt to distinguish and evaluate the impacts of forest certification on the environmental characteristics of regeneration microsites should involve a wider variety of environmental variables, specifically those related to soil conditions. Although the potential to reduce the changes in adult tree composition and soil conditions through the use of RIL is relatively well established in experimental studies, the impact of certified logging on the microsites that act as 'regeneration units' has been little studied. The method of analysing the MN and MS gap characteristics separately could be further developed in empirical studies. By combining these kinds of studies with information on actual harvesting intensities, important knowledge could be provided on the mechanisms of certification standards and other similar instruments in improving ecological sustainability.

Finally, the need for ecological studies that concentrate on the impact of on-the-ground logging operations should be emphasized. Typically, ecological studies are conducted in controlled field experiments or protected forests. With certified forestry, such studies overlook the combined ecological impacts of past resource use, improved control over illegal logging, the dissemination of knowledge about conservation needs and environmentally sound logging practices, and the benefits of reduced forest clearance. Furthermore, as the results of ecological forest research are often poorly communicated to the forest decision-makers (Guariguata and Pinard, 1998), forest certification could potentially offer a good forum for channelling the results of current ecological research to practical forestry.

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Appendix: List of studied species. Key to abbreviations: T = timber tree species, NT = non-timber tree species, LD = light-demanding, ST = shade-tolerant, (int.) = intermediate (tolerates shaded conditions reasonably well).

Species	Family	Timber /non-timber	Light ecology	References	Papers
Alchornea latifolia Sw.	Euphorbiaceae	NT	LD	Dalling and Tanner, 1995; Finegan and Delgado, 2000; Myster and Walker, 1997; Schierenbeck et al., 1997	II
Ampelocera hottlei (Standl.) Standl.	Ulmaceae	NT	ST	Martínez-Garza et al., 2005	III
Asterogyne martiana (H. Wendl) H. Wendl ex Hemsl.	Arecaceae	NT	ST	(Chazdon, 1985)	III
Billia hippocastanum Peyr.	Hippocastanaceae	NT			II
Brosimum alicastrum Sw.	Moraceae	T	ST	Benítez-Malvido and Lemus-Albor, 2005; Montgomery and Chazdon, 2001; Rocas, 2003	I, II, III
Calathea sp. G. Mey.	Marantaceae	NT	LD	Costa et al., 2002	III
Callicarpa acuminata Kunth	Verbenaceae	NT			II
Calophyllum brasiliense Cambess.	Clusiaceae	T	ST (int.)	Carpenter et al., 2004; Finegan and Delgado, 2000; Flores, 1994; Loik and Holl, 1999	I, II, III
Calyptranthes chytraculia (L.) Sw.	Myrtaceae	NT			II
Carpotroche platyptera Pittier	Flacourtiaceae	NT			II
Cecropia peltata L.	Cecropiaceae	NT	LD	Dalling and Hubbell, 2002; Quintana-Ascencio et al., 1996	II
Cedrela odorata L.	Meliaceae	T	LD	Martínez-Garza, 2003	III
Cephaelis Sw sp. (= Psychotria L. sp.)	Rubiaceae	NT	LD		II
Cephaelis tomentosa (Aubl.) Vahl (= *Psychotria poeppigiana* Müll. Arg.)	Rubiaceae	NT	LD	Hartshorn and Poveda, 1991	II
Cestrum racemosum Ruiz & Pav.	Solanaceae	NT			II
Cojoba arborea (L.) Britton & Rose	Fabaceae	T	ST	Benítez-Malvido and Lemus-Albor, 2005; Sandí and Flores, 2003	I, II, III
Cordia alliodora (Ruiz & Pav.) Oken	Boraginaceae	T	LD	Dalling et al. 2004; Finegan and Delgado, 2000; Guariguata, 2000; Martínez-Garza, 2003; Menalled and Kelty, 2001	II

Cupania guatemalensis (Turcz.) Radlk.	Sapindaceae	NT			II
Dendropanax arboreus (L.) Decne. & Planch.	Araliaceae	NT			II
Dialium guianense (Aubl.) Sandwith (= *Arouna guianensis* Aubl.)	Fabaceae	T	ST	Benítez-Malvido and Lemus-Albor, 2005; Pennington and Sarukhan, 1968	I, II, III
Dichapetalum bullatum Standl. & Steyerm.	Dichapetalaceae	NT			II
Eugenia L. sp.	Myrtaceae	NT			II
Garcinia intermedia (Pittier) Hammel	Clusiaceae	NT		Martínez-Garza, 2003	II, III
Guarea grandifolia DC.	Meliaceae	T	ST	Flores, 2003a	I, II
Guarea macrophylla Vahl	Meliaceae				II
Guarea sp. F. Allam. ex L.	Meliaceae	T	ST		III
Hamelia patens Jacq.	Rubiaceae	NT (shrub)			II
Heliocarpus appendiculatus Turcz.	Tiliaceae	NT	LD	Guadarrama et al., 2004; Martínez-Garza, 2003; Tinoco-Ojanguren and Pearcy, 1995; Quintana-Ascencio et al. 1996	I, II
Hernandia stenura Standl.	Hernandiaceae	NT			III
Huertea cubensis Griseb.	Staphyleaceae	T	ST	Anon., 1992	I, II
Hyeronima alchorneoides Allemão	Euphorbiaceae	T	ST (int.)	Balderrama and Chazdon, 2005; Clark and Clark, 1999; Gerwing, 1995; Menalled and Kelty, 2001; Flores 2003b	I, II, III
Inga Mill. spp,	Fabaceae	NT			II
Licania platypus (Hemsl.) Fritsch	Chrysobalanaceae	NT			II
Licaria capitata (Schltdl. & Cham.) Kosterm.	Lauraceae	NT			II
Licaria Aubl. spp.	Lauraceae	NT			II
Lonchocarpus guatemalensis Benth.	Fabaceae	NT			II
Matayba oppositifolia (A. Rich.) Britton	Sapindaceae	NT		Cordero and Boshier, 2004; Hartshorn and Poveda, 1991	II
Miconia Ruiz & Pav. sp.	Melastomataceae	NT			II
Mortoniodendron vestitum Lundell	Tiliaceae	T	LD		I, II
Ochroma pyramidale (Cav. ex Lam.) Urb.	Bombacaceae	NT	LD	Dalling et al., 1999	II
Pachira aquatica Aubl.	Bombacaceae	NT			II
Pausandra trianae (Müll. Arg.) Baill.	Euphorbiaceae	NT			II
Pentagonia macrophylla Benth.	Rubiaceae	NT			II
Perymenium grande Hemsl.	Asteraceae	NT	LD	Cordero and Boshier, 2004; Kass and Somarriba, 1999	II

Piper auritum Kunth	Piperaceae	NT	LD	Burger, 1991, Quintana-Ascencio et al., 1996; Vazquez-Yanes and Orozco-Segovia, 1992	II
Piper L. spp.	Piperaceae	NT	LD		II
Piper aduncum L.	Piperaceae	NT	LD	FAO, 1976, Quintana-Ascencio et al., 1996; Rogers and Hartemink, 2000	III
Piper umbellatum L.	Piperaceae	NT			II
Pourouma sp. Aubl.	Cecropiaceae	NT			III
Pouteria izabalensis (Standl.) Baehni	Sapotaceae	NT			III
Pouteria sp. Aubl.	Sapotaceae	NT			III
Protium sp. Burm. f. SYN. *Icica* Aubl.	Burseraceae	NT			III
Pterocarpus rohrii Vahl	Fabaceae	NT			II
Pterocarpus sp. Jacq.	Fabaceae	NT			III
Rinorea hummelii Sprague	Violaceae	NT			II
Schizolobium parahyba (Vell.) S.F. Blake	Fabaceae	NT			II
Simarouba glauca DC. (syn. *Simarouba amara*)	Simaroubacea e	T	ST (int.)	Clark and Clark, 1999; Guariguata, 2000	I, II
Sloanea sp. L.	Elaeocarpacea e	NT			III
Swietenia macrophylla King	Meliaceae	T	LD		III
Terminalia Amazonia (J.F. Gmel.) Exell	Combretaceae	T	LD (int.)	Carpenter et al., 2004; Flores, 1994; Flores, 2003c; Pennington and Sarukhan, 1968	II
Trema micrantha (L.) Blume	Ulmaceae	NT	LD	Dalling and Hubbell, 2002; Hooper et al., 2002; Quintana-Ascencio et al., 1996	II
Trichospermum sp. Blume	Tiliaceae	NT			II, III
Reinhardtia gracilis (H. Wendl) Drude ex Dammer	Arecaceae	NT	ST		III
Vatairea lundellii (Standl.) Killip ex Record	Fabaceae	NT			III
Virola koschnyi Warb.	Myristicaceae	T	ST	Balderrama and Chazdon 2005; Benítez-Malvido and Lemus-Albor, 2005; Finegan and Delgado, 2000; Flores, 2003d; Montgomery and Chazdon, 2001	I, II, III
Vismia guianensis (Aubl.) Pers.	Clusiaceae	NT	LD	Dias-Filho and Dawson, 1995	II
Vismia baccifera (L.) Triana & Planch.	Clusiaceae	NT	LD		II
Vochysia guatemalensis Donn. Sm. (= *Vochysia hondurensis* Sprague)	Vochysiaceae	T	LD (int.)	Balderrama and Chazdon, 2005	III

Xylopia frutescens Aubl.	Annonaceae	NT	LD	Zahawi and Augspurger, 2006	II
Zanthoxylum sp. L.	Rutaceae	NT			III
Total species:	Total families				