Community-based mangrove management in West Mexico: Assessing the sustainability of small-scale selective wood harvesting

by

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Montréal, Québec, Canada
June 2017

A thesis submitted to McGill University in partial fulfillment of the requirements of the degree of Master of Science

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Abstract

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Mangroves are highly productive and biodiversity-rich socio-ecological ecosystems that provide vital goods and services to millions of people, including wood, a renewable natural capital, which is the primary source of energy and construction material for several coastal communities in developing countries. Unfortunately, mangrove loss and degradation occur at alarming rates. In some regions, unregulated and unsustainable mangrove wood harvesting are important causes of degradation. Community-based harvesting is a common practice but few successful case studies are known and studies evaluating its sustainability, and effect on different ecosystem services are lacking. Therefore, my research explores the sustainability of regulated community-based small-scale mangrove timber production by assessing its effect on multiple forest structure attributes, such as tree density, natural regeneration, and wood volume.

For this study, I identified and selected the most intensively and recently harvested and the most conserved mangrove natural stands within a Management Unit for Wildlife Conservation in West Mexico, where local communities have been managing mangroves for decades for both domestic and commercial purposes. In contrast to mangrove over-harvesting, industrial and illegal logging scenarios elsewhere authorized community-based forestry activities in the area follow a unique approach including four management units: conservation, wood production, protection, and restoration. Average tree density and number of shared diameter size classes were not significantly different between harvested and non-harvested stands. Diameter-size class analysis revealed a good representation of different tree development stages from young to mature trees in both conditions. Height class-analysis showed that average natural regeneration of seedlings was similar in both forest conditions suggesting that regulated selective harvesting does not hinder the natural regeneration of mangrove forests in the Management Unit. Although average wood volume between conditions was significantly different, high volumes of wood were recorded in both harvested and non-harvested stands. These results indicate that community-based mangrove wood harvesting may contribute to enhancing the establishment of seedlings, securing wood stocks in the long-term and preserving landscape connectivity.

Mangroves in the studied area are likely to be resilient to wood harvesting and hurricanes, as average tree densities and natural regeneration estimated were comparable to well-managed mangrove forests in other regions of the world. Consequently, the spatial and temporal trade-offs of mangrove harvesting and the provision of multiple ecosystem services may be minimal, as selective tree harvesting is conducted only in small production areas allowing the maintenance of canopy cover, along with roots, soil, and biodiversity conservation. Overall, my research findings suggest that community-based mangrove forestry through Management Units for Wildlife Conservation could be a cost-effective scheme to manage and conserve mangrove forests and their ecosystem services within and beyond protected areas, while providing local livelihoods and helping reduce illegal logging. If implemented with a multidisciplinary perspective that incorporates scientific assessments this win-win strategy may contribute to achieving national and international environmental and sustainable development agreements that could provide multiple social, ecological and economic benefits from local to global scales. Including the protection of traditional knowledge, biodiversity, and renewable natural capital, as well as forest-based climate change mitigation and adaptation.

Résumé

Gestion communautaire des mangroves de l'ouest du Mexique: évaluation de la durabilité de la récolte sélective du bois à petite échelle

Les mangroves sont des écosystèmes socio-écologiques très productifs et riches en biodiversité. Ils fournissent des services écosystémiques essentiels à des millions de personnes, notamment à travers le bois, un capital naturel renouvelable, qui est la principale source d'énergie et de matériaux de construction pour plusieurs communautés côtières dans les pays en développement. Pourtant, la perte et la détérioration des mangroves se produisent à un rythme alarmant. Dans certaines régions, l'exploitation non réglementée et non durable du bois de mangrove est ainsi une des causes importantes de la dégradation de cet écosystème. Malgré le fait que la récolte communautaire est une pratique courante, peu d'études de cas réussis sont connues et des études évaluant la viabilité, la durabilité et l'effet sur les différents services écosystémiques font défaut. Ma recherche porte sur la durabilité des récoltes de mangrove réglementées par les communautés locales en évaluant leurs effets sur plusieurs attributs de la structure de la forêt, dont la densité des arbres, la régénération naturelle et le volume du bois.

Pour cette étude, j'ai sélectionné parmi les forêts de mangroves situées dans une unité de gestion et conservation de la faune de l'ouest du Mexique, les zones les plus intensivement et récemment récoltées de même que les plus conservées, dans une région où les communautés locales ont géré la production de bois de mangroves à des fins domestiques et commerciales pendant des décennies. Contrairement à des scénarios de surexploitation ou d'exploitation illégale ou industrielle de mangroves observés ailleurs, les activités forestières dans cette région sont soumises à une réglementation qui intègre de manière unique quatre volets de gestion: la production, la conservation, la protection et la restauration. La densité moyenne des arbres et le nombre de classes de taille de diamètre partagé n'étaient pas significativement différents entre zones de mangroves récoltées et non récoltées. L'analyse de la structure des classes de diamètre a révélé une bonne représentation des différents stades de développement d'arbres jeunes à mâtures dans les deux conditions. L'analyse de catégories de hauteur a montré que la régénération naturelle moyenne des semis était aussi similaire dans les deux conditions forestières, ce qui indique que la récolte sélective réglementée n'empêche pas la régénération naturelle des peuplements de forêts de mangroves dans l'unité de gestion. Bien que le volume moyen du bois entre les conditions soit significativement différent, des volumes élevés ont été enregistrés dans les peuplements récoltés et non récoltés. Ces résultats indiquent que la récolte communautaire de bois de mangrove peut contribuer à améliorer l'établissement de jeunes arbres, à sécuriser les stocks de bois à long terme et à préserver la connectivité du paysage.

Les mangroves dans la zone étudiée semblent résilientes à la fois à la récolte du bois et aux ouragans, puisque les densités moyennes d'arbres et la régénération naturelle estimées étaient comparables à des mangroves bien gérées dans d'autres régions du monde. Par conséquent, les compromis spatio-temporels de la récolte de mangroves et la fourniture de services écosystémiques multiples sont minimes, car la récolte sélective d'arbres ne se fait que dans de petites zones de production permettant le maintien de la couverture de la canopée, de même que la conservation des racines, du sol et de la biodiversité. Dans l'ensemble, mes résultats de recherche suggèrent que la foresterie communautaire des mangroves à travers des unités de gestion pour la conservation de la faune pourrait être un système rentable pour gérer et conserver les forêts de mangrove et leurs services écosystémiques à l'intérieur et à l'extérieur des aires protégées, tout en fournissant des moyens de subsistance aux communautés locales et à réduire

le risque de récolte illégale. Si elle est mise en œuvre avec une perspective pluridisciplinaire qui intègre des évaluations scientifiques, cette stratégie avantageuse peut contribuer à la réalisation d'accords nationaux et internationaux en matière d'environnement et de développement durable qui pourraient offrir de multiples avantages écologiques, sociaux, et économiques à des échelles locales et globales, notamment en favorisant la protection des connaissances traditionnelles, de la biodiversité et du capital naturel renouvelable, ainsi qu'en termes d'atténuation et d'adaptation des changements climatiques fondées sur les forêts.

Acknowledgements

I wish to thank all the people who provided advice, feedback and guidance throughout my studies and research work—this project wouldn't be possible without your collaboration.

First, I would like to express my deepest gratitude to my Supervisor, Dr. Gail L. Chmura, for giving me the opportunity to develop an independent research project. I am thankful for her advice, guidance and enthusiastic encouragement. I greatly enjoyed our long talks about similarities and differences between salt marshes and mangroves! I am very grateful to my Committee Members, Dr. Margaret Kalacska and Dr. Tim Moore, for their support and feedback. I would also like to extend my sincere appreciation to my advisor from Mexico, Dr. Juan Ignacio Valdez Hernández, for his useful contributions. His input and encouragement were invaluable to the completion of this project.

My fieldwork could not have been possible without the help of many people. I am especially grateful to the community "Ejido San Blas" who allowed me to conduct my research in their Management Unit "UMA", for providing me with the boat transportation and for the exhaustive fieldwork collaboration. Furthermore, I would like to express my thanks to Ing. Alfonso Solis for sharing critical information and key insights on different sites for my research, in addition to providing geographical information data and helping to organize the fieldwork agenda with the community. Many thanks to my three field assistants from the community for their technical assistance and key contributions. I appreciate all the vital help you offered during my fieldwork—especially on those hot, wet, muddy, mosquito-infested days. Thank you for sharing your knowledge and for your sincere interest in my work.

I am grateful to my friend, Alberto López Carbajal, for accepting the challenge to accompany me during my fieldwork. I will continue to cherish our memories of those days. Many thanks to Dr. Kalacska for sharing her insights into fieldwork research and for providing me with a laptop and software to locate my sites.

I would like to thank my colleagues, friends, and professors from the University of Guadalajara for their support. In particular, I would like to thank Eduardo J., Dr. Eduardo R., and José Luis N, as well as to Lupita for sharing fieldwork equipment and instrumental literature. In 2006, I was invited to participate in a training course on mangrove forestry and monitoring for local communities in Nayarit, West Mexico where I learned part of the methodology applied in this research and about the mangrove community-based work in the area. I am very thankful to the organizers and lecturers for the opportunity to discover a world that was previously unknown to me.

I would like to acknowledge all the organizations that kindly granted me financial awards to pursue my master studies and to conduct my independent research study. First and foremost, I would like to thank my home country, Mexico, for granting me a generous Master's Scholarship of the Mexican National Council for Science and Technology (CONACyT). My graduate studies and research also received financial support from the Russell E. Train Education for Nature Program (WWF-EFN) Fellowship and Research Award, for which I am very grateful. Specifically, I would like to extend my sincerest appreciation to the late Mr. Russell E. Train for being a constant source of inspiration to advance the study of nature and its conservation alongside local communities. I am honored to have been granted a Fellowship in his name. I would like to thank the Department of Geography at McGill University, specifically for the Graduate Enrolment and Principal's Fellowships that I received, as well as to the Warren Development Fieldwork Award that founded part of my fieldwork.

Many thanks to all the people who provided feedback on my abstracts and posters that were presented at international conferences. Special thanks to Dr. Chmura, Dr. Valdez-Hernández, Juan P., and Niki K. The results of this study have been presented in diversified approaches in the following international meetings:

-3rd Science for Biodiversity Forum 'Mainstreaming biodiversity for well-being: contributions from science to policy' - UN Biodiversity Conference/COP13, Cancun, Mexico.

-53rd Meeting of the Association for Tropical Biology and Conservation Organization: Reconciling Conservation and Sustainable Use of Biodiversity, Montpellier, France.

-Biodiversity, Ecosystem Services and Sustainability, McGill University, Canada.

-IUCN World Parks Congress - Parks, people, planet: inspiring solutions, Australia.

-ITTO/FAO/FONAFIFO's International Forum on Payments for Environmental Services of Tropical Forests, San Jose, Costa Rica.

-Forests as Capital: International Society of Tropical Foresters, Yale University, USA.

-50th Meeting of the Association for Tropical Biology and Conservation Organization, “New Frontiers in Tropical Biology: The Next 50 Years”, San Jose, Costa Rica.

Finally, I would like to express my profound gratitude to my family and friends. Your support and motivation have been my anchor along the way. To my husband and son, no words can describe my gratitude. I am very thankful to the both of you for your endless patience, fierce encouragement, and love—especially during those long work hours into the night when it was most needed. Also, a much-deserved thanks to the newest member of our family. Most importantly, however, I would like to thank my husband. Your resilience and hard-work meant that I could focus on my dreams and continue to delve into my passion. You have been a pillar throughout this project and I am forever grateful for your boundless support. We did it!

Last but not least, a special thanks to my brother and sister for their continued support throughout my graduate studies. To my mother, who taught me the importance of education and perseverance, I thank you. Your love and support gave me the strength to pursue my dreams.

To all the mangrove guardians “local communities and indigenous peoples” in Mexico
Thanks for all that I have learned from you and for being a source of inspiration!

¡Muchas gracias! Merci! Thank you!

Chapter 1. Introduction

This first Chapter provides the introduction, justification and objective of my research.

1.1. Introduction

Achieving a fair balance between the conservation and sustainable use of forests is one of the most important and challenging sustainable development goals to safeguard local livelihoods along with the various vital goods and services that forests provide on local to global scales. Mangroves, the intertidal evergreen forests that grow along the coasts of the subtropics and tropics are valuable multi-use socio-ecological systems as they provide a full range of direct and indirect ecosystem services critical for the survival of mangrove-dependent communities (Costanza *et al.* 1997; Ewel *et al.* 1998; Rönnbäck 1999; Walters *et al.* 2008; Barbier *et al.* 2011; UNEP 2014; Lee *et al.* 2014). For example, they provide ecosystem services with global benefits, such as carbon storage (Chmura *et al.* 2003) and habitat for migratory birds, as well as coastal protection. Their support of fish and shrimp production has direct economic benefit regionally and locally (Rönnbäck 1999; Nagelkerken *et al.* 2008). One provisioning service that mangrove forests provide at local scales is wood, a renewable natural capital, which is the primary source of energy and construction material of many rural communities and indigenous peoples living on the coasts of developing countries.

Mangroves cover less than 1% of the world's forested areas but are disappearing at higher rates than any other tropical forests (Duke *et al.* 2007). Major anthropogenic drivers of direct loss are land use changes to unsustainable shrimp culture, agriculture (e.g. Valderrama *et al.* 2014), coastal development, and coastal squeeze (Valiela *et al.* 2001; Alongi 2002; Duke *et al.* 2007; FAO 2007). Natural drivers of mangrove loss and degradation and modification of forest structure include larger-scale hurricanes and sea level rise (e.g. Cintron *et al.* 1978;

Cahoon *et al.* 2003; Gilman *et al.* 2008). However, anthropogenic hydrological modifications due to the construction of dams, artificial inlets, and roads, as well as river diversion for agriculture are the principal cause of mangrove loss and degradation in several regions, such as in West Mexico. In many cases, local and regional mangrove loss and degradation cause irreversible damage to the functioning of the ecosystem and the provision of ecosystem services affecting local communities' traditional livelihoods and their customary rights to ancestral land.

Nevertheless, in some regions, unregulated and unsustainable mangrove wood harvesting is an important driver of habitat and biodiversity loss, as well as mangrove degradation. Mangrove overharvesting and clearing are considered important drivers of mangrove degradation in several American, African and Asian countries (Valiela *et al.* 2001; Alongi 2002; Saenger 2002; Walters 2005b; Din *et al.* 2008). Alternatively, conserving, restoring and managing mangrove forests sustainably at watershed scales can play a significant role in climate change mitigation strategies to reduce CO_2 emissions (e.g. Laffoley and Grimsditch 2009; Nellemann *et al.* 2009), to protect the coast against hurricanes and to provide sustainable livelihoods.

Community-based mangrove wood harvesting has a long history in tropical developing countries for local use and commercial purposes (e.g. Saenger 2002), a practice that is conducted mainly following a selective small-scale approach (Walters *et al.* 2008). While most unregulated small-scale harvesting is considered unsustainable causing important changes to mangrove forest structure, studies have shown that authorized mangrove harvesting causes minimal changes (e.g. Valdez-Hernández 2002a). Mexico is well-known for the multiple successful examples of community-based forest management (e.g. Merino *et al.* 1997); however, they are predominantly documented in terrestrial forests and little is known about community-based management in mangrove forests. Mexico ranks fifth among the 17-megadiverse countries that host more than

70% of earth's biological diversity (Mittermeier and Goettsch 1992; Sarukhán *et al.* 2009; Llorente-Bousquets and Ocegueda 2008). The majority of Mexico's forests are in the hands of communities. "Ejidos" and indigenous communities legally own 74% of the biodiversity and 67% of the coastland (SEDATU 2012), including mangrove forests. Local and indigenous communities are key actors in the effective implementation and permanence of mangrove conservation and sustainable forest management policies and strategies. A good mangrove forestry example is the Marismas Nacionales-San Blas Ecoregion (66,020 ha of mangrove forests), in the State of Nayarit, West Mexico, where local communities have managed mangroves for decades for small-scale selective wood harvesting, as well as conserving and restoring them (e.g. Fajardo 2007), but only a few studies have evaluated the effects of wood production on mangrove forest structure (Valdez-Hernández 2002a).

For this study, I selected the Management Unit for Wildlife Conservation (UMA) of the community Ejido San Blas, where regulated community-based mangrove harvesting activities have occurred since the 1990s following traditional ecological knowledge. Resulting in one of the most conserved mangrove areas in the southern part of Marismas Nacionales-San Blas Ecoregion, West Mexico. The Ejido San Blas was a pioneer in the sustainable management of coastal resources for their livelihoods and in regulating mangrove forestry activities (e.g. Valdez-Hernández 2004). Since the late 1920's, mangrove forests in the area have been harvested for commercial purposes, but their domestic use may date back to pre-Hispanic times (Fajardo 2014). Mangrove forests in the UMA were severely damaged by the impact of Hurricane Kenna in 2002. Therefore, this study examines the sustainability of regulated small-scale selective community-based harvesting of *Laguncularia* in the UMA of the Ejido San Blas and the socio-ecological resilience and productivity of mangrove forests.

1.2. Research Objective and Questions

My research focuses on evaluating the sustainability of regulated community-based small-scale selective mangrove timber production by assessing its effect on provisioning services "raw materials – wood" and multiple forest structure attributes eight years after the impact of a large-scale hurricane. Documenting and communicating management and harvesting approaches conducted in natural mangrove forests by local communities and indigenous people and evaluating their effects on provisioning ecosystem services and mangrove structure will provide critical information for the conservation and sustainable management of mangroves ecosystems in the long-term. This information could provide the basis for the development of criteria and indicators that can be used by decision-makers to draft and implement forestry and public policies that integrate community-based mangrove conservation and sustainable forest management, as well as payments for ecosystem services. Therefore, my study aims to contribute to filling the gaps in the documentation and evaluation of successful sustainable community-based mangrove management. Specifically, I examine the sustainability and effects of regulated small-scale selective harvesting on several forest structure attributes including stem density, natural regeneration, canopy cover and standing commercial wood volume. I assessed and compared these attributes between the most recently harvested and the most conserved mangroves forests of the Management Unit for Wildlife Conservation located within the San Blas Mangrove System, West Mexico. My primary hypothesis is that community-based small-scale selective mangrove harvesting as in the Management Unit "UMA" of the Ejido San Blas does not reduce the capacity of mangrove forests to regenerate, to allow the maintenance of sustainable yields and landscape connectivity. I further hypothesize that tree density and natural regeneration are higher in harvested stands as a result of *Laguncularia* coppicing and the creation of temporary small canopy gaps after harvesting. Consequently, I hypothesize that standing wood volumes are similar in harvested and non-harvested forests, as only a few trees of commercial size are removed selectively by hand in harvested stands. Lastly, I hypothesize that mangrove cover has been maintained in the Management Unit "UMA" over the period 1970/1980 to 2010. Hence, for this study I address the following questions:

1) Is forest structure in harvested mangrove stands different from non-harvested stands?

2) What are the wood volumes in harvested and non-harvested mangrove forests? Are wood volumes in harvested stands large?

3) Has small-scale harvesting hindered natural regeneration of San Blas Mangrove Forests within the Management Unit? Is natural regeneration higher in harvested stands? Does natural regeneration come from seeds or vegetative reproduction?

4) Is mangrove wood production by the Ejido San Blas sustainable? What are the keys factors to the success of this community-based case study and its implications for sustainable mangrove management?

5) Has wood production reduced mangrove connectivity in the San Blas Mangrove System? Has wood production reduced mangrove cover in the last 30-40 years (period 1970/1980 to 2010)?

6) Could mangrove wood production through Community-based Management Units for Wildlife Conservation be an effective conservation scheme?

Chapter 2. Theoretical Framework

This second Chapter summarizes the relevant theoretical framework related to sustainable forest management, sustainable livelihoods, ecosystem services, community forestry, mangrove ecology, and silviculture.

2.1. Sustainability - Sustainable Development

Taking into account that the main objective of this study is to evaluate the sustainability of community-based mangrove harvesting, it is necessary first to define the term "sustainable", as it has many different meanings (Hopwood *et al.* 2005). The 1987 Brundtland Report, "Our Common Future", contains one of the most commonly used definitions of sustainable development: "development which meets the needs of the present without compromising the ability of future generations to meet their own needs" (WCED 1987). In general terms, sustainability is a state or process maintained indefinitely, conceived as the limitless balanced interaction of three different elements or dimensions: environmental, economic and societal (O'Riordan 1998; Giddings *et al.* 2002). The UN Development Programme's Human Development Report 2011 further links sustainability, social justice and greater access to a better quality of life. Scoones 1998, proposed a framework under the Sustainable Livelihoods Program of the Institute of Development Studies (IDS) to define sustainable rural livelihoods, which is frequently used as a reference. Under this scheme, livelihoods are divided into five categories: natural capital, economic/financial capital, human capital, social capital, and others.

Sustainable forest management applies the sustainable development concept to forestry. Probably the most widely used definition of sustainable forest management is the one drafted by the United Nations "a dynamic and evolving concept that aims to maintain and enhance the economic, social and environmental value of all types of forests, for the benefit of present and future generations" (www.fao.org). A United Nations Resolution adopted in 2007 (UN 2008, Resolution 62/98), recognizes seven elements that characterize sustainable forest management: 1) extent of forest resources; 2) forest biological diversity; 3) forest health and vitality; 4) productive functions of forest resources; 5) protective functions of forest resources; 6) socio-economic functions of forests; and 7) legal, policy and institutional framework" (SCBD 2009). The UN Food and Agriculture Organization (FAO) considers sustainable forest management as: "The stewardship and use of forests and forest lands in a way, and at a rate, that maintains their biodiversity, productivity, regeneration capacity, vitality and their potential to fulfill, now and in the future, relevant ecological, economic and social functions, at local, national, and global levels, and that does not cause damage to other ecosystems". The Society of American Foresters refers to sustainable forestry as: "The practice of meeting the forest resource needs and values of the present without compromising the similar capability of future generations. It involves practicing a land stewardship ethic that integrates reforestation, managing, growing, nurturing, and harvesting trees for useful products with conservation of soil, air and water quality, wildlife and fish habitat, and aesthetics" (Helms 1998). However, it lacks the inclusion of social aspects, considering that forests are complex socio-ecological systems (e.g. Cote and Nightingale 2012; Puettmann *et al.* 2013).

2.2. Natural Capital and Ecosystem Services

In recent times forest conservation is given more attention for the critical role of forest in providing both ecosystem services and natural capital to achieve sustainable development and to mitigate climate change. "Natural capital" refers to Earth's biosphere stocks of renewable and non-renewable biotic/living resources produced by ecological systems, including oxygen, water, carbon reservoirs, land, and wood, which contribute to the generation of goods and services (e.g. Jansson *et al.* 1994; Costanza *et al.* 1997). To achieve the sustainable use of natural resources a minimum requirement is the maintenance and enhancement of renewable biotic natural capital stocks (Costanza and Daly 1992). The accurate estimation and valuation of natural capital are critical to the sustainability of the ecological economic system as it generates a significant part of the Biosphere's goods and services (Costanza and Daly 1992).

Ecosystem services provide valuable direct and indirect ecological, economic, and social goods and services critical for both the functioning of the planet and human well-being (Costanza *et al.* 1997, Daily *et al.* 2000). The term ecosystem services refer to the tangible (material) and intangible (non-material) direct or indirect benefits that humans obtain from natural ecosystem processes, some times referred as goods (e.g. food, wood, etc.) and services (e.g. waste assimilation, climate regulation, soil formation) (Costanza *et al.*1997; MEA 2005). Daily 1997, defined ecosystem services as the "conditions and processes through which natural ecosystems, and the species that make them up, sustain and fulfill human life".

Ecosystem services categorization occurs in multiple ways, one of the most used is the one proposed by the Millennium Ecosystem Assessment (MEA 2005). Applying functional criteria, the MEA classified ecosystem services into four categories: 1) provisioning services, which are related to natural capital stocks, such as food, water, medicines, and wood 2) regulating services, which correspond to the benefits obtained from regulation of ecosystem processes, and which contribute to coastal protection, flood prevention, soil erosion, and climate and water regulation, 3) cultural services are non-material benefits that directly affect people, such as traditional beliefs, cultural heritage, recreational and landscape services and 4) supporting services, are all the services needed to maintain the other services and the conditions for life on Planet Earth, such as nutrient cycling, soil formation, and primarily production.

2.3. Silviculture Practices for Natural Forest Management

Forest management refers to "the process comprising the set of actions and procedures aimed at the management, cultivation, protection, conservation, restoration and use of resources and environmental services of a forest ecosystem, considering ecological principles, respecting the functional integrity and interdependence of resources and without reducing the productive capacity of ecosystems and resources therein" (LGDFS 2016).

In the context of this study, natural forest management refers to the management of forests that have regenerated naturally (www.fao.org). Forests that have been planted, such as commercial plantations, are not considered natural forests. Natural silviculture is "the practice of controlling the establishment, growth, composition, health and quality of natural forests to meet diverse needs and values" (www.fao.org). A silviculture approach is defined by the "interventions applied to forests to maintain or enhance their utility for specific purposes, such as the production of wood and other forest products, biodiversity conservation, recreation and the provision of environmental services" (www.fao.org). The conservation and sustainable management of natural forests contribute to the preservation of biological diversity (SCDB 2009). For instance, mangrove forests help maintain ecological dynamics critical for the conservation of endangered biological diversity, and the provision of multiple ecosystem services (FAO 2015). A critical issue is how a managed forest will be maintained in the long-term while conserving their ecological functions. Thus, establishing how the forest will regenerate is the basis for a sustainable forest management.

In silviculture, the management of natural forests can be broadly categorized using natural regeneration approaches as even-aged and uneven-aged forests or as high and low recovery systems (Wittwer *et al.* 1990; Kuuluvainen *et al.* 2012). An example of a silviculture approach is the selective harvesting of single trees for wood production and its recovery by natural regeneration.

2.3.1. Natural Regeneration

There are several natural regeneration techniques used in forestry depending whether the objective of the forest management plan is to obtain an even or uneven-aged forest. Wittwer *et al.* (1990) have summarized the different recovery methods used in even-aged (i.e. clearcutting, seed tree, and shelterwood methods) and uneven-aged forests (i.e. selection with two approaches: single-tree and group selection).

Regeneration methods used in uneven-aged stands include vegetative sprouting from existing root systems, shoots from the cut stems (coppice shoots), and natural regeneration from seeds. Uneven-aged methods promote the presence of several tree age classes within forest stands. These methods can be applied at small scales to reduce logging impacts. If conducted using adequate principles, controlled and regulated harvesting does not affect natural forest regeneration. On the contrary, it could contribute to the establishment of fast growth and woody pioneer species that otherwise could be lower in non-harvested forests (Duah-Gyamfi *et al.* 2014). Studies focused on modeling and structural dynamics of uneven-aged forest stands are numerous for terrestrial forests (e.g. Haight 1985; O'Hara and Gersonde 2004; Pukkala *et al.* 2009, 2010), but less abundant for mangrove forests.

2.3.2 Small-scale Selective Harvesting

In an uneven-aged stand, selective harvesting or cutting can be applied to remove only trees of selected species and diameter size classes authorized for domestic or commercial proposes while maintaining landscape connectivity, canopy cover, and habitat for biodiversity. Small-scale selective harvesting has been proposed as a solution to promote the sustainable use of forests and a better forestry practice in comparison to clear cutting approaches (e.g. SEMARNAT 2012). The sustainability of selective harvesting depends on, in part, whether the logging management approach applied has a high-impact or reduced-impact (e.g. Cazzolla-Gatti *et al.* 2015). Selective harvesting has ecological effects that vary with the management and harvesting approaches, e.g., the number of trees cut per unit area, the diameter of the trees logged, species harvested, and wood volume harvested. For instance, in terrestrial tropical forests, selective logging could be conducted at different scales either with the use of traditional tools such as machetes or axes, or mechanized with the use of bulldozers and electrical tools, which can result in extensive damage to the environment, such as soil compaction and decreased natural regeneration (Chazdon *et al.* 2007). Selective harvesting could also have impacts on biodiversity conservation (e.g. Johns 1988), which tend to be site and species specific (e.g. Bourque and Villard 2001), but studies evaluating the effects of logging on ecosystem services are often lacking.

2.4. Mangrove Ecosystems

Mangroves are composed of evergreen woody halophytes (salt-tolerant plants) distributed along coasts of estuaries and coastal lagoons in the tropics and subtropics (Pool *et al.* 1977). The term "mangrove" can refer to either the ecosystem or to individual plants (Lugo and Snedaker 1974; Tomlinson 1986). Mangrove ecosystems have ecological connectivity with nearby ecosystems, such as seagrass, salt marsh, coral reef and freshwater wetlands (e.g. Bacon and Alleng 1992). They are recognized for the provision of long-term multiple ecosystem services, natural capital, biodiversity conservation and socioeconomic benefits for the survival of millions of mangrove-dependent communities and indigenous peoples (e.g. Walters *et al.* 2008; UNEP 2014).

Globally there are between 50 to 75 mangrove species distributed in 20-23 genera belonging to 16 families (Blasco 1984a; Tomlinson 1986), which present various adaptations such as aerial roots, pneumatophores, viviparous embryos, floating propagules and salt excretion glands that enable their survival in saline, flooded, muddy and shifting coastal conditions. Mangrove understory's can include a variety of biota such as ferns, algae, lichens, and fungi (Blasco 1984a).

2.4.1. Factors Influencing Mangrove Forest Structure

Several internal-external abiotic and biotic factors can affect mangrove forest structure (Blasco 1984b; Tomlinson 1986; Lovelock *et al.* 2005; Berger *et al.* 2008). These factors include substrate type, sedimentation rates, soil conditions, geomorphology, micro-topography, surface elevation, nutrient availability, light, salinity, weather, propagule dispersal and establishment, herbivory and faunal activity within the forest floor, as well as coastal and oceanic dynamics, including marine currents and winds (e.g. Lugo and Sneadeker 1974; Cintron *et al.* 1978; Zimmermann and Thom 1982; Thom 1982; Cintron and Schaeffer-Novelli 1984; Tomlinson 1986; Smith 1987, 1992; McKee 1993; Lovelock *et al.* 2005; Sengupta *et al.* 2005; Cannicci *et al.* 2008; Krauss *et al.* 2008; Fromard and Proisy 2010; Feller *et al.* 2010; Walcker *et al.* 2015). Interspecific competition is considered an important factor in determining mangrove species diversity and distribution patterns (Thom 1982). Anthropogenic disturbances, such as mangrove wood harvesting and large-scale hurricanes are considered to cause a great disturbance in mangroves, influencing temporal and spatial forest structure (e.g. Roth 1992; Sherman *et al.* 2000). The magnitude and periodicity of such factors and the frequency of storms are important (Lugo and Sneadeker 1974; Cintron and Schaeffer-Novelli 1984; Smith *et al.* 1994, 2009; Milbrandt *et al.* 2006).

There are two common assumptions about mangroves relevant to forest structure patterns. The first is that mangroves are distributed in monospecific bands parallel to water bodies, as a result of their physiological adaptations to varied hydroperiods and salinities. Alternatively it is assumed that this zonation of species represents their succession in time (Tomlinson 1986). Multiple studies have focused on determining the existence of mangrove species zonation patterns with mixed results (e.g. Chapman 1976; Semeniuk 1980; Ruwa 1993; Ellison and Farnsworth 1993; Matthijs *et al.* 1999; Ellison *et al.* 2000; Castañeda-Moya *et al.* 2006; Fickert and Grüninger 2010). Mangrove ecological patterns are frequently explained by physicochemical gradients resulting from a combination of several biotic and abiotic environmental factors within the intertidal zones. Intertidal wetlands, such as mangroves, are transitional zones between marine, estuarine and terrestrial wetlands or other terrestrial ecosystems, and changes in the environment from land to saltwater create gradients or zones in which ecological processes diversify (Ruwa 1996). Hydrologic processes in mangrove ecosystems are one of the primary limiting factors, including fresh and seawater flows; wave intensity; tidal frequency and amplitude; precipitation; evapotranspiration; and water table dynamics (Twilley and Chen 1998; Whelan *et al.* 2005; Krauss *et al.* 2006).

2.4.2. Mangrove Ecosystem Services

Mangroves, one of the most highly productive and biodiversity-rich ecosystems in the world, provide various vital ecosystem services from local to global scales. The economic, ecological and social importance of mangrove ecosystems have only recently began to be recognized (e.g. Nagelkerken *et al.* 2008) as earlier research mainly focused on studying their unique ecological processes and physiological adaptations to grow in saline environments.

The ecosystem services that mangroves provide stem from them being both wetlands and forests. They exchange organic matter and sediments with adjacent ecosystems and with coastal waters (Kristensen *et al.* 2008). Mangroves contribute to climate regulation by influencing local and regional temperature and precipitation. They are important sinks for carbon dioxide (Chmura *et al.* 2003), captured through photosynthesis and stored as above and belowground biomass. Mangroves play an important role in soil formation and retention; they are critically important in the cycling and storage of nutrients, pollutants, and terrestrial particulate organic matter. They provide essential habitat for terrestrial and marine endangered species (Ewel *et al.* 1998; Nagelkerken *et al.* 2008). Finally, mangroves play a significant role in protecting the coast from, flooding and coastal erosion associated with hurricanes and tsunamis (Rönnbäck 1999; Badola and Hussain 2005).

From a socioeconomic perspective, mangroves are important coastal ecosystems through their provisioning services that sustain local livelihoods. They are a source of food, supporting fish and shrimp production (Ewel *et al.* 1998; Rönnbäck 1999), with not only a direct economic benefit at local scales but with a contribution to global food security. The indirect relationship between mangroves and fish productivity is well recognized. It has been estimated that between 70 to 90% of the fish populations worldwide depend on mangroves during at least one or more stages of their life cycle (e.g. Hamilton *et al.* 1989). They provide breeding, nursery and feeding habitat and refuge for many commercially exploited marine organisms and endangered species. Mangroves are a source of medicines and other biochemical products such as tannins.

One long-recognized provisioning service of mangrove forests is wood, a renewable natural capital, which is the primary source of energy for many rural communities and indigenous people in developing countries. There are a vast number of forest products that have significant uses for the daily living of people, including tannin, construction timber, and charcoal (Lugo and Snedaker 1974). In developing countries, many communities build their houses, construct boats and fish traps from mangrove wood.

Mangroves are regarded as one of the most important socio-cultural ecosystems in tropical areas, as they provide multiple cultural services related to spiritual and religious aspects linked to the services they provide. In many countries in the Americas, pre-Hispanic indigenous people developed their cultures near mangrove ecosystems. Nowadays, many rural communities and indigenous peoples still live near or within mangrove areas in developing countries. Mangroves are multi-use ecosystems: for recreation, ecotourism, research and education activities. Regrettably, mangroves are disappearing and being degraded at rapid rates (Valiela *et al.* 2001; Alongi 2002; Duke *et al.* 2007), limiting their capacity to provide ecosystems services (Costanza *et al.* 1997; Ewel *et al.* 1998; Barbier *et al.* 2011; Lee *et al.* 2014), as both wetlands and forests. It is considered that about 35 to 86% of mangroves worldwide have been lost mainly by unsustainable shrimp culture activities in developing countries (Valiela *et al.* 2001; Alongi 2002), affecting local livelihoods (e.g. Duke *et al.* 2007).

2.4.3. Mangrove Silviculture

Silviculture and forestry management approaches vary widely among mangrove systems and regions, as do the methods to evaluate mangrove forest structure (Table 1). While some harvesting and wood production practices are unsustainable others may present an opportunity for the conservation of multiple ecosystem services provided by mangrove ecosystems. Mangrove silviculture have a long history in developing countries, but most of the widely spread reports on these are based on activities in African and Asian countries (e.g. Saenger 2002; Walters *et al.* 2008). Less is known about mangrove silviculture practices and successful management in the Americas, in some cases due to limited access to information or due to language barriers. As in other tropical forests, forestry management methods include clear cutting, large-scale commercial harvesting or small-scale selective harvesting (Walters *et al.* 2008). There are also different social organization schemes to produce mangrove wood. Mangrove wood production could be divided into three categories: government-based as in Malaysia, foreign/industrial-based as occurred in many South Asian countries in the past centuries (e.g. Sukardjo 1987), or local community-based as in some Latin American countries (Valdez-Hernández 2002a, 2004) and African countries.

The Sundarbans, the world's most extensive mangroves (600,000 ha) located in India and Bangladesh have the longest recorded history of management for timber harvesting, since 1769, with forest plans prepared in 1893-1894 (in Kairo *et al.* 2001). In Sumatra, wood production for commercial purposes started in the 16th century (Sukardjo 1987). Matang forest in Malaysia (40,000 ha) have been commercially harvested since the early 1900s (Gong and Ong 1995). In the case of Mexico, the Marismas Nacionales-San Blas Ecoregion (66,020 ha) has been harvested since the 1920s for commercial purposes, but their use for domestic purposes may date to pre-Hispanic times (Fajardo 2014).

Table 1. Methodologies used to evaluate mangrove forest structure and natural regeneration in the world. DBH = Diameter at breast height.

Country/ Region	Sampling unit size and intensity	Sampling location	Juveniles	Tree		Reference
			Height (cm)	DBH (cm)		
Mexico	Trees/Stems: Twenty 300 m^2 (30 x 10 m) plots, each divided into three 100 m^2 subplots Saplings: Two 16 m^2 subplots (4 x 4 m) Seedlings: Five 1 m^2 subplots	Plots randomly established on forest stands harvested and non-harvested	Class I: < 30 Class II: > 30 to 130 Class III: > 130	<2.5	>2.5	Valdez-Hernández 2002a
	Trees/Stems: Eighteen 6 m radius circular units of 113 m^2, six units located in each zone Saplings: 16 m^2 sub-units Seedlings: Five 1 m^2 quadrats	Sampling units located in three zones parallel to the course of river along a gradient of soil moisture.	<130	-	-	Rocha-González *et al.* 2012
Federated States of Micronesia	1 m^2 plots (Six to 118 per site)	One sampling unit of 2 ha established in three mangrove stands	Class I: <50 Class II: >50 to 100 Class III: >100 to 150 Class IV: 150 to 200	-	-	Pinzon *et al.* 2003
Dominican Republic	23 permanent sampling units (size not specified) Seedlings: Twelve 1 m^2 plots	Sampling units established along two transects	Seedlings: ≤100 Saplings: >100	<5	>5	Sherman *et al.* 2000
Mozambique & Tanzania	Vegetation plots 10 x 10 m. Juveniles were also counted within the 100 m^2 plots.	Plots located considering the upper landward zone, middle zone and lower seaward zone of the mangrove system	Class I: <40 Class II: >40 to 150 Class III: >150 to 300	<2.5	>2.5	Bandeira *et al.* 2009
Kenya	Up to 106, 100 m^2 (10 x 10 m) plots per site	Plots established along belt transects of 10 m width perpendicular and parallel to the creek across the forest	Class I: < 40 Class II: >40 to 150 Class III: >150 to 300	<2.5	>2.5	Mohamed *et al.* 2009
	Between 31 and 60, 10 x 10 m^2 plots per site				>2.5	Kairo *et al.* 2002a

2.4.3.1. Sustainability of Mangrove Harvesting

Various scientists have questioned the sustainability of mangrove wood production, but mainly evaluating unregulated small-scale practices by local communities, and less is known on the impact of clear cut and large-scale and mangrove harvesting. Alongi and de Carvalho (2008) reported that small-scale mangrove harvesting could result in unnoticeable degradation, including the reduction of aboveground biomass and an increase in canopy gaps. However, long-term studies are needed to adequately evaluate the effects of harvesting on the entire suite of ecosystem services. Tovilla-Hernández *et. al.* 2001 conducted a six-year study to evaluate the effects of mangrove cover loss due to harvesting. They found that soil temperature and salinity increased, decreasing soil fertility and concluded that mangrove deforestation affected organic matter production, biogeochemical and biological cycles, as well as natural regeneration. In southern Thailand, mangrove harvesting follows strip clear-felling methods, which Alongi (2009) reports as unregulated and unsustainable.

2.4.3.2. Large-scale and Clear Cut Mangrove Wood Production

Worldwide there have been situations in which mangroves have been overharvested due to unsustainable large-scale commercial and industrial exploitation (e.g. Sukardjo 1987). In the past, concessions of mangrove land to foreign private companies resulted in over extraction of wood in an unsustainable manner and degradation of mangrove ecosystems, mainly in Asian and African countries (Sukardjo 1987). The Matang Mangrove Forest Reserve (MMFR) in Malaysia is recognized as a case of successful mangrove forest management (Gong and Ong 1995; Saenger 2002; Alongi 2002; Alongi 2009; UNEP 2014). Matang commercial harvest is regulated, government-based and large-scale (Saenger 2002), following a clear-cutting method where a few trees are left as parent trees to promote forest regeneration (Alongi 2009). However, the sustainability of this method has been questioned due to the removal of seedlings and saplings limiting the natural regeneration and resilience of the forest, as well as its ecological functionality (e.g. Gong and Ong 1995; Goessens *et al.* 2014). Only 20% of the area is left as reserve forest (Alongi 2009). In recent years manual planting has been required (Alongi 2009). Besides, the removal of so many trees can impact forest biodiversity (e.g. Khaleghizadeh *et al.* 2014) and landscape connectivity. Gong and Ong (1995) reported that in a 30-year rotation period, the Matang forest productivity declined more than 50%.

2.4.3.3. Small-scale and Selective Community-based Mangrove Forestry

Community-based mangrove wood harvesting for both local use and commercial purposes has a long history in tropical developing countries (e.g., Saenger 2002; Valdez-Hernández 2002a, 2004; Walters *et al.* 2008). The production of mangrove wood by local communities and indigenous people frequently follows a small-scale harvesting approach (e.g. Alongi and de Carvalho 2008; Walters *et al*. 2008; Valdez-Hernández 2002a, 2004). However, most of their practices are unregulated, as they use mangrove wood mainly for domestic proposes. Unsustainable small-scale harvesting has been frequently identified as an important driver of mangrove degradation. Studies have reported small-scale unregulated and unsustainable practices in the Philippines (Walters 2005a, b), Timor-Leste (Alongi and de Carvalho 2008), Mozambique and Tanzania (Bandeira *et al*. 2009), and in Guerrero, Mexico (Tovilla-Hernández *et al.* 2001). Kairo *et al*. 2002a reported unregulated mangrove harvesting in Kenya that resulted in the decline of the commercially valued mangrove species. In the State of Nayarit on the west coast of Mexico, local communities situated within the Marismas Nacionales-San Blas Ecoregion follow small-scale selective harvesting (Valdez-Hernández 2002a). Valdez-Hernández (2002a) conducted a study in this region in the Ejido Villa Juárez and concluded that mangrove forests structure was similar in harvested and non-harvested stands in community-based managed forests for regulated selective harvesting.

The most widespread and well-documented case studies of mangrove wood production in the world are government or industrial-based in Asian countries (e.g. Saenger 2002; UNEP 2014). Community-based mangrove harvesting is not as well documented, studied or regulated and documented examples of sustainable community-based harvesting are rare. Also few studies have evaluated the ecological effects of small-scale logging and its sustainability. In some tropical countries, such as in Mexico, the misconception that community-based mangrove harvesting is unsustainable has resulted in hands-off conservation schemes in some regions while in others it is recognized and regulated, such as in the State of Nayarit (SEMARNAT 2012).

2.5. Mexican Mangrove Forests

Mexico has the fourth largest mangrove area in the world with 775,555 ha (2015), which accounts for more than 5% of the global cover (FAO 2007; Spalding *et al.* 2010; Giri *et al.* 2011; Troche-Souza *et al.* 2016). The majority of Mexico's mangroves are located within federal and state protected areas (465,333 ha = 63%), and 59 Ramsar Sites (Troche-Souza *et al.* 2016). Mangroves are distributed in 17 littoral States along the three coasts of Mexico: Pacific, Gulf of Mexico and Caribbean (Figure 1). Mexican mangrove communities are composed of three main true mangrove species: *Rhizophora mangle L.* (Red Mangrove), *Avicennia germinans* (L.) L. (Black mangrove) and *Laguncularia racemosa* (L.) (White mangrove), with the secondary presence of *Conocarpus erectus* (Miranda and Hernández 1963; Flores *et al.* 1971; Pool *et al.* 1977; Tomlinson 1986; Pennington and Sarukhán 2005). Two other mangrove species are found in South Mexico: *Rhizophora Harrisonii*, a natural hybrid of *R. mangle* and *R. racemosa* (Cornejo 2013), and *Avicennia bicolour*, a species generally restricted to the Pacific Coast of Central America and southern Mexico (Jiménez 1990).

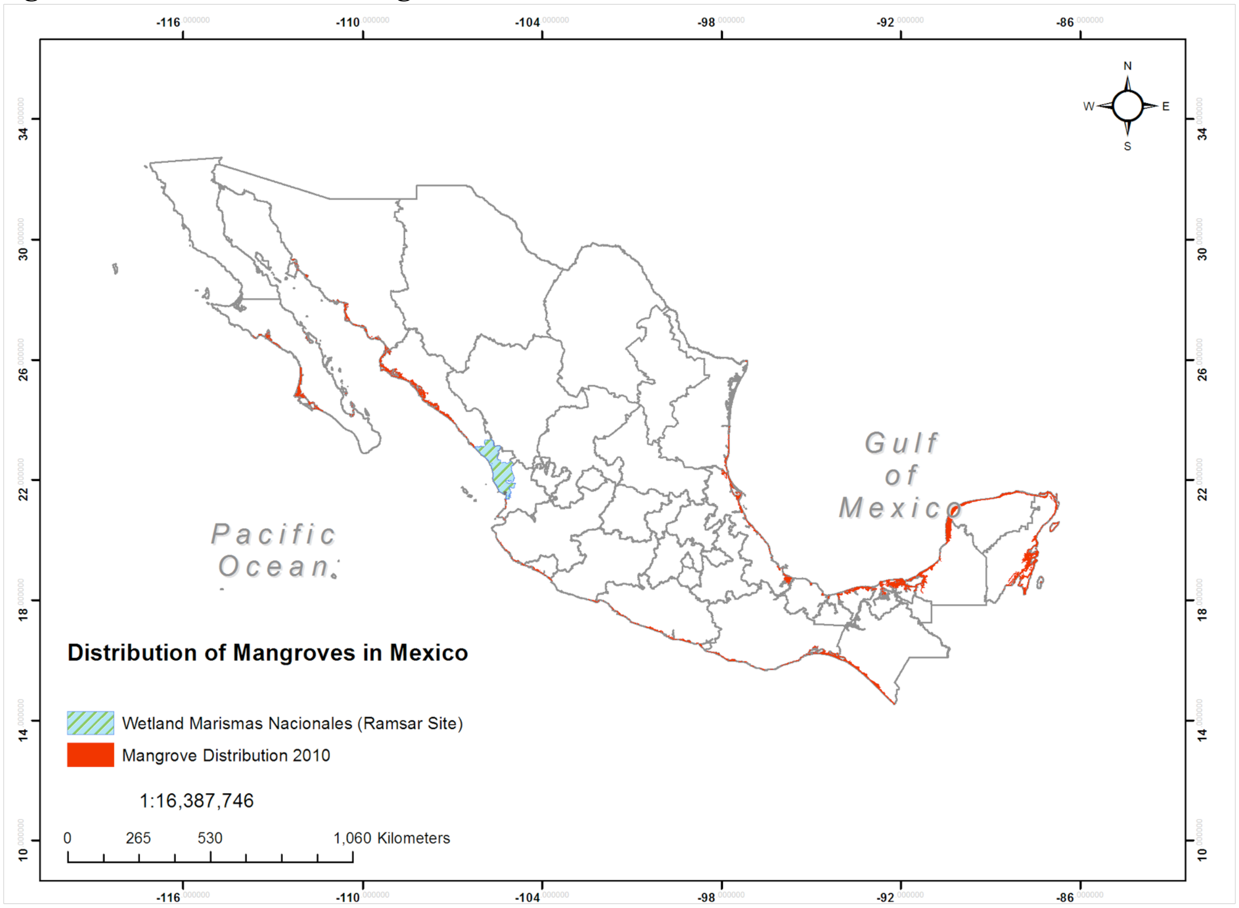
2.6. Mangrove Conservation and Sustainable Forest Management in Mexico

Mangroves are one of the most protected and monitored ecosystems in Mexico, as several regulations have been developed for the conservation of these ecosystems (NOM-022-SEMARNAT-2003) and their species (NOM-059-SEMARNAT-2010) (www.dof.gob.mx). The four most common mangrove species have been officially protected since 1994 (Table 2), and their conservation, restoration, and sustainable use have been subject to the Mexican General Wildlife Law since 2000. Law bans mangrove clear-cut, and only few communities hold permits for small-scale wood production. In 1997, the Mexican Government established a System of Management Units for Wildlife Conservation (UMAS, acronym in Spanish) regulated by the Ministry of Environment (SEMARNAT) to strengthen biodiversity conservation, socio-economic development and the sustainable use of natural resources based on production needs in the rural sector (www.semarnat.gob.mx). The objective of the Management Units is to promote alternative means of production with the rational and planned use of renewable resources based on Management Plans.

Table 2. Historical changes in the protection of mangrove species in Mexico.

Family	Species	Category	Year	Category	Year	Category	Year
Combretaceae Combretaceae Rhizophoraceae Verbenaceae	*Laguncularia racemosa* *Conocarpus erectus* *Rhizophora mangle* *Avicennia germinans*	Subject to special protection	1994	Subject to special protection	2002	Threatened	2010

Figure 1. Distribution of mangroves in Mexico.



2.7. Community-based Land Ownership in Mexico: Ejido (socio-communal landholdings)

The majority of Mexico's forests are in the hands of local communities "Ejidos" and "Comunidades". The "Ejido" is a unique socio-communal land ownership system, and is the primary land tenure scheme in Mexico. It incorporates communal land ownership with personal use. The Ejido concept originated after the Mexican Revolution (1910-1917) from the expropriation of land from large private owners "Hacendatarios". The land expropriated was redistributed to groups of landless peasants "Campesinos" during a period of seven decades. In 1992, the Article 27 of the Mexican Constitution was modified, giving the ejidos and other communities the legal ownership of their lands comprising water bodies, forest and other natural resources.

“Communidades” is another communal social landholding and in general terms refers to the ancestral land occupied and used by indigenous peoples. Currently, between 68.6 and 78.4 million hectares of forested land equivalent to between 70 to 80% of Mexico’s forests are in the hands of “Ejidos” and “Comunidades” (estimated from SEDATU 2012 data). It is estimated that 50% of Mexico’s territory is forested land represented by multiple ecosystems, which include mangrove forests. A high percentage of Mexicans from diverse ethno-cultural origins living in rural and coastal areas are highly dependent on forest for their survival and their livelihoods.

Chapter 3. Methods

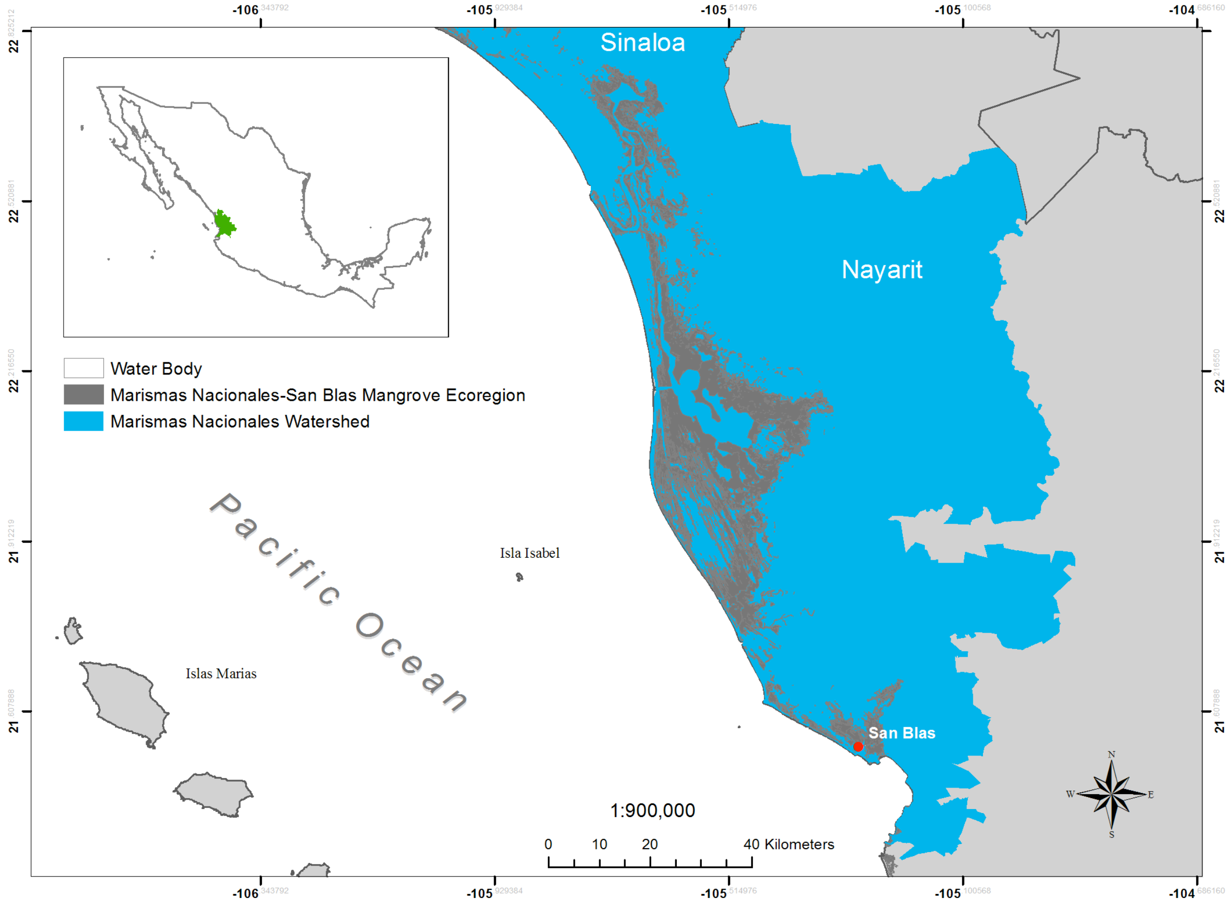
Chapter 3 summarizes the relevant methodological framework applied to select the study area, design the semi-structured sampling, and the justification of the methods used to determine mangrove forest structure. It also provides a description of the studied area.

3.1. Study Area San Blas Mangrove System

The study area is located in the southern part of the San Blas Mangrove System and Marismas Nacionales-San Blas Ecoregion in the State of Nayarit, West Mexico (Figure 2), one the most productive, biodiversity-rich and extensive mangrove areas on the Pacific coast of the Americas (Flores-Verdugo *et al.* 1990), designated as a Ramsar Site in 1995 (Ramsar 2009). Here mangroves can extend up to 17.2 km landward from the coast along the Huicicila-San Blas River (Figure 4). The climate in the San Blas region is predominately tropical sub-humid A(w), with a mean temperature of 25.7°C, and an annual precipitation of 1,459 mm, which corresponds to the tropical dry forest ecosystems according to Holdridge's life zones. The rainy season is from June to October, which corresponds to the hurricane season.

The San Blas Mangrove System is part of the San Cristobal sub-basin of the Huicicila-San Blas watershed (344,300 ha) located in the Huicicila Hydrological Region (RH13 of CNA). The system also receives fresh water from the biggest river on the Mexican Pacific, the Lerma-Santiago River (RH12), mainly during flooding periods and through groundwater springs and rainfall. Huicicila-San Blas is a sinuous single-channel river with meandering currents and dendritic drainage that flows the coastal alluvial plains of San Blas. The San Blas Mangrove System is also located within a priority hydrological region for the Mexican National Commission for the Knowledge and Use of Biodiversity (CONABIO) (Arriaga *et al.* 2002).

Figure 2. Study area location map, Southern Marismas Nacionales-San Blas Ecoregion, State of Nayarit, West Mexico.



San Blas is one of the most conserved mangrove areas in the region, providing multiple ecosystem services, such as ecotourism activities and biodiversity conservation. The San Blas Mangrove system supports a high diversity of fauna. It is part of a biological corridor for migratory birds that arrive from Canada, the United States, and northern Mexican areas in winter. As many as 252 species of birds have been reported in the study area, of which it is estimated that 40% are migratory, representing one of the areas of highest concentration of migratory birds in North America (Canada-USA-Mexico). Approximately 36 bird species are endemic to the region (CONABIO 1999; WWF 2001), making San Blas a high-priority area for biodiversity conservation.

The flora of San Blas Mangrove System is among the richest in the Marismas Nacionales-San Blas Ecoregion with 42 species of vascular plants registered (Valdez-Hernández 2002b). The mangrove vegetation is composed of three mangrove species: *Laguncularia racemosa* (L.) Gaertn. f., *Avicennia germinans* (L.) Stearn and *Rhizophora mangle* as well as the mangrove associate *Conocarpus erectus* (Valdez-Hernández 2004). *Laguncularia* is the dominant mangrove species in the Marismas Nacionales-San Blas Mangrove System (Pool *et al.* 1977). All three mangrove species are used locally for domestic purposes. Wood obtained from *Avicennia* and *Rhizophora* is used as construction material for houses, fishing facilities and restaurants, as well as in medicine (e.g. Kovacs 1999). *Laguncularia* wood is used to smoke fish for traditional local dishes, as a building material (i.e. stakes and poles) for agriculture purposes, fences and walls, and in artisanal fishing (Kovacs 1999). Other wood uses in the area include charcoal and firewood (Pool *et al*. 1977).

The majority of the human settlements near the mangrove are small rural communities. El Puerto de San Blas, a fishing village of about 10,187 habitants, is an exception. The principal local livelihoods include agriculture, cattle, artisanal fishing, ecotourism, and mangrove wood production. According to the National Institute of Statistics and Geography (INEGI 2012), the study sites are located on land designated for agriculture and forestry activities. The study area is vulnerable to natural disturbances, including flooding, tropical storms and hurricanes. Mangrove forests in the area have experienced major hurricanes and storms in the past.

In 2002, Hurricane Kenna, a category 4 hurricane, was the last significant natural phenomenon that impacted the local mangrove forests. It made landfall near the village of San Blas with average winds of 230 km hr^{-1}, and up to 275 km hr^{-1} (SMN/CNA 2002). It was accompanied by a storm surge of 4.0 to 5.5 m, and an eyewall of 19 km. The hurricane caused the loss 40-60% of leaf cover and tree damage as illustrated in Figure 3 (*personal communication with local villagers*).

Figure 3. Tree damage due to Hurricane Kenna 2002 in a non-harvested stand.

3.1.1. Study Sites Location

The study sites are located in an area known as Singaita, 5 km inland from the Pacific coast within the Management Unit for Wildlife Conservation (UMA) of the Ejido San Blas (Figure 4). Mangroves within the UMA reach 8 km inland. The mangrove forest stands are situated in middle coastal plains in relation to the coast seashore and upper landward areas with significant inputs of freshwater during rainy season, and with daily tidal influence (Figure 4). Mangrove forests structural attributes correspond to well-developed Riverine and Basin Forests that occur along river and creek drainages regularly flooded by daily tides. Mature forests can develop large and dense canopy depending on climate, topography and on the effect of anthropogenic activities (Tomlinson 1986). Local communities in the area are the legal owners of mangrove land, which have been conserving, restoring and managing mangroves throughout decades for small-scale selective wood harvesting, used both for domestic and commercial purposes. The community Ejido San Blas by their initiative designated areas for mangrove conservation 30 years ago, which has resulted as one of the most conserved mangrove areas in the State of Nayarit. The community has followed forest management plans since the 1990s and in 1998 registered a Management Unit for Wildlife Conservation (UMAs) regulated by the Minister of Environment (SEMARNAT, acronym in Spanish), which is divided into four conservation areas: conservation, restoration, production, and protection (Valdez-Hernández 2004; SEMARNAT 2012) (Figure 4, Table 3).

Table 3. Description of the four mangrove management areas under the Management Unit for Wildlife Conservation of the Ejido San Blas in the State of Nayarit, West Mexico.

Management Units for Wildlife Conservation			
Protection Zone	**Production Zone**	**Conservation Zone**	**Restoration Zone**
Areas where mangroves have the ecological function of protecting rivers, estuaries, and lagoons. It covers the vegetation that goes up to 10 to 25 m inland.	Areas where individuals of a dominant species are likely to be used for commercial purposes. The size of the areas designated for small-scale wood harvesting may vary according to wood volume at local scales and the required rotation estimated from forestry assessments.	Areas designated exclusively for habitat conservation of flora and fauna included in the Mexican Regulation NOM-059-SEMARNAT 2010, as endangered, threatened and subject to special protection.	Areas designated for activities aimed at restoring and rehabilitating degraded forest ecosystems, to restore ecosystem functions, and to maintain favorable conditions for ecosystems persistence and evolution.

3.2. Research Design and Field Sampling

This study focused on stands dominated by *Laguncularia*, as wood production is limited to the harvesting of this mangrove species. Selection criteria included: 1) 5 km distance from the coast to reduce natural variability due to tidal gradients, 2) intensity of harvesting and 3) age of trees. To evaluate the effects of small-scale harvesting on mangrove forest structure two stands were compared, one was identified as the most intensively and continuously harvested with the presence of 25-year-old trees and juveniles and the second was non-harvested "old-growth" forest with mature and 50 years-old trees present.

A total of six 300 m^2 (30 x 10 m) plots were established within harvested and non-harvested sites. Three plots were located 30 m apart in each site in low (20 m inland), mid (60 m inland) and high (100 m inland) intertidal zones (Figure 5). Plots were situated with their long axis parallel to creeks. Each plot was divided in three 100 m^2 (10 x 10 m) subplots. Information on the management and harvesting approach conducted by the Ejido San Blas was obtained through the key informant technique as suggested by Tremblay (1957).

Figure 4. Zonation of the UMA of the Ejido San Blas. Sampling sites location: A) harvested and B) non-harvested forest.

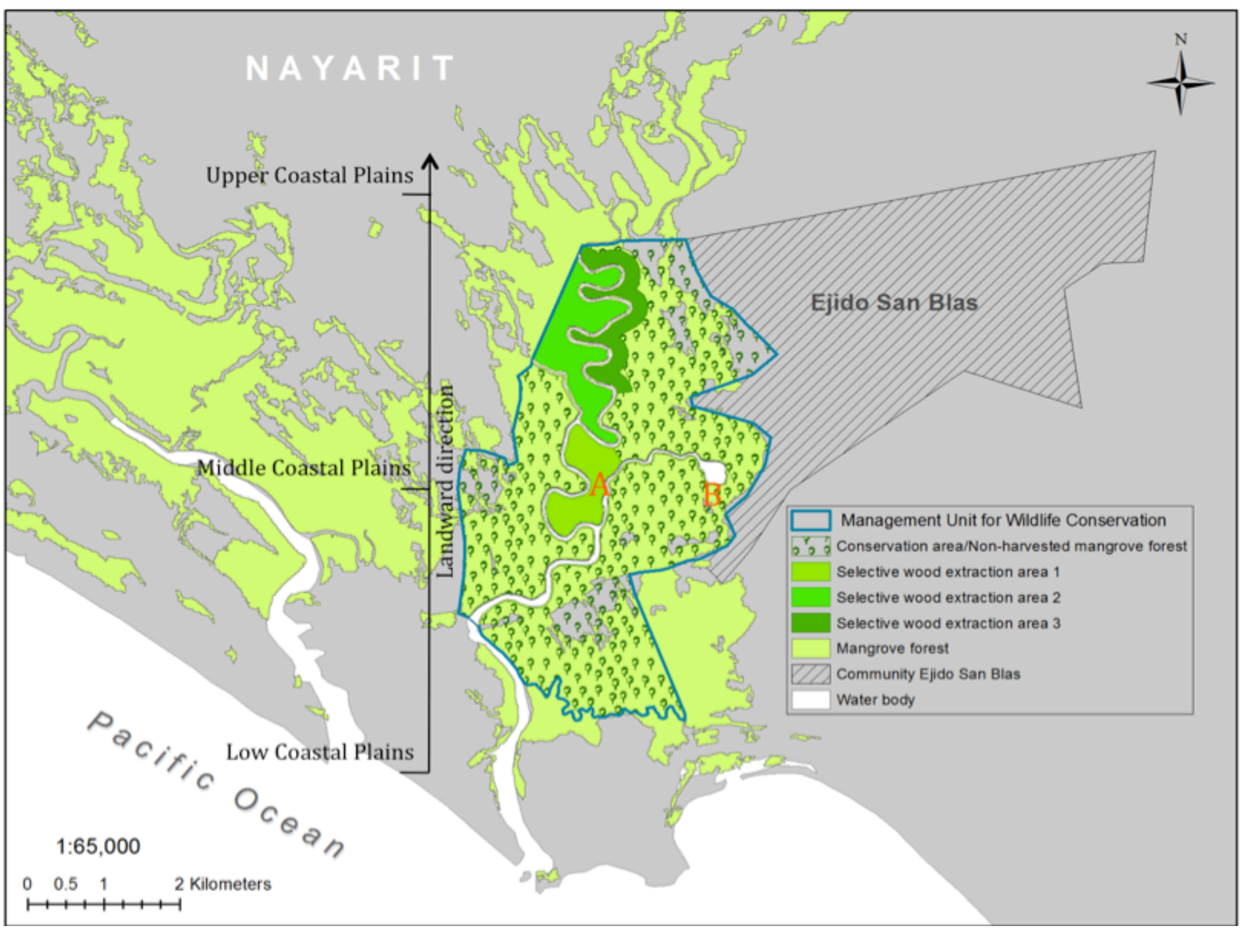
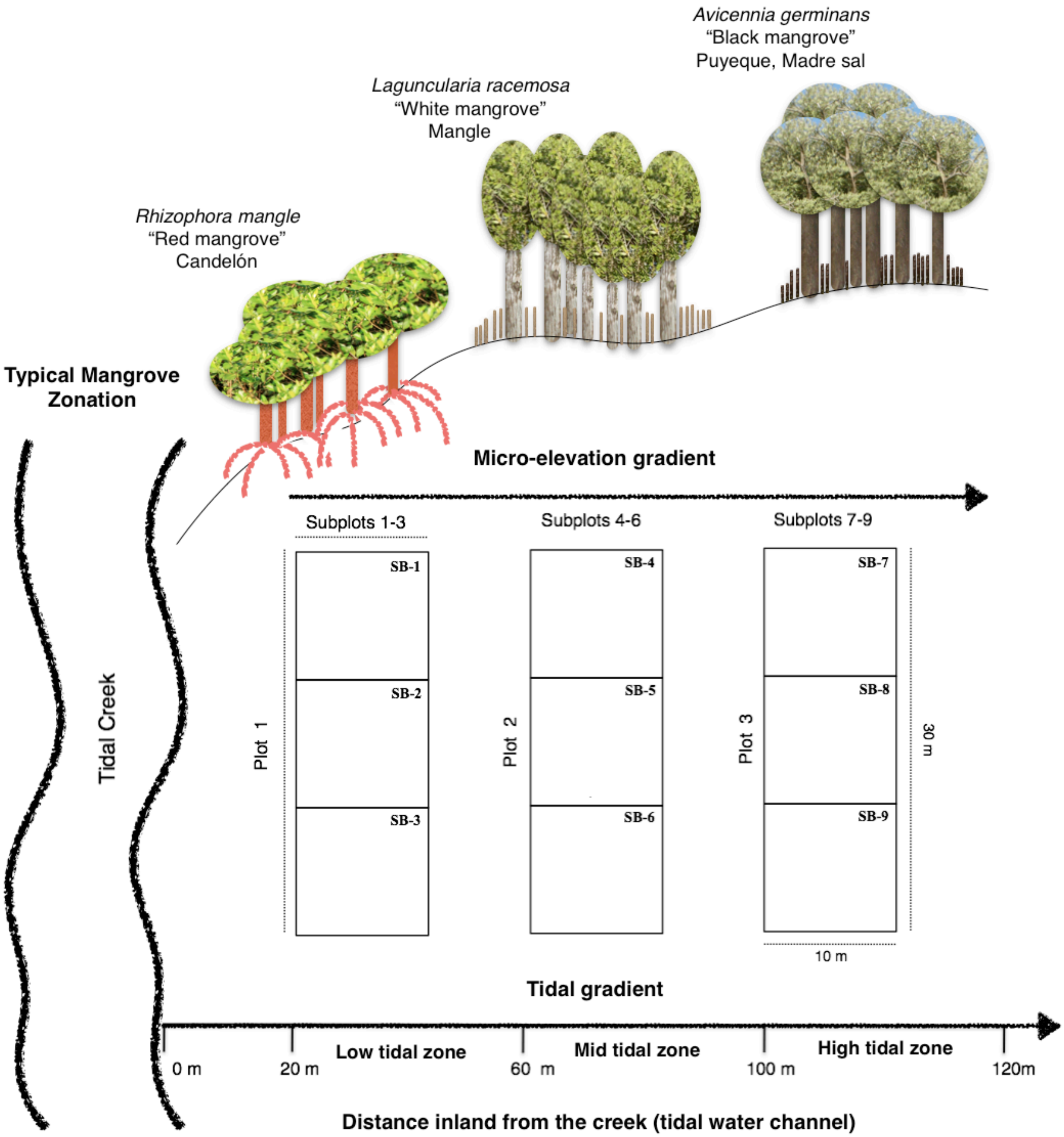


Figure 5. Sampling design.



3.3. Interstitial Salinity and pH

Interstitial water was obtained from perforated PVC pipes (e.g. Serrano-Díaz *et al.* 1995; Febles-Patron *et al.* 2009) with a 10.16 cm diameter that were installed to 60 cm depth within each 300 m^2 plot in low, mid and high intertidal zones. In the lowermost 50 cm of each tube 0.5 cm diameter holes were placed. A couple of hours after installation soil interstitial water was suctioned from each pipe using a small hose, and stored in plastic containers. pH was measured using a potentiometer and salinity using a refractometer. Water was collected two times during the raining season with an interval of two months.

3.4. Micro-topography

Micro-topography surveys were conducted at each site using the hose level method (e.g. García-Márquez 1994; Andrade and Ferreira 2006). The hose level method is based on the physical principle of communicating vessels. As gravity and hydrostatic pressure are constant throughout the hose, water will settle at the same level regardless of the elevation of each extreme. To measure the micro-elevation the water level difference between the two stakes is estimated, which provides information on the change in elevation from point one to point two in centimeters. I constructed a micro-topography surveying instrument to achieve accurate measurements, which consisted of two stakes, to which a measuring tape was attached. The extremes of a 20-m long flexible transparent hose filled with water were attached to each stake. Micro-topography was evaluated along a 120 m transect perpendicular to the creek. Readings were taken every 13 m along the 120 m transect. Before each reading, the hose was checked for bubbles.

3.5. Mangrove Forest Structure

Forest structure was determined applying methods similar to those used in previous studies in the area (Valdez-Hernández 2002a) and elsewhere for mangrove ecosystems (Pool *et al.* 1977) with the collaboration of local villagers previously trained as communal forestry technicians. Diameter and height of all trees with a diameter at breast height (DBH) > 2.5 cm were recorded in each 100 m^2 subplot. For *Laguncularia* and *Avicennia* tree diameter was measured at 1.3 m from the ground, and for *Rhizophora,* the diameter was measured at 30 cm above the highest prop root. In the case of *Laguncularia* individual stems in a clump were counted. Stems were measured and counted if their centers were within the unit boundary. In each 100 m^2 subplot, the % canopy cover was estimated along a 10 m transect. Relative tree dominance was determined by dividing the number of trees of each species by the total number of trees (Cintrón and Schaeffer-Novelli 1984).

3.5.1. Natural Regeneration

Within each 300 m^2 plot, natural regeneration was analyzed by documenting seedlings and saplings in two subplot sizes randomly located. Seedlings were measured within five 1 m^2 square subplots. Within two 25 m^2 subplots all saplings (<2.5 cm of diameter and >1.3 m tall for *Laguncularia* and *Avicennia* or 1.5 m tall for *Rhizophora*) were measured. Three classes were used to categorize natural regeneration: Class I = *Laguncularia* and *Avicennia* seedlings shorter than 30 cm, and *Rhizophora* seedlings shorter than 50 cm; Class II = *Laguncularia* and *Avicennia* seedlings taller than 30 cm but shorter than 1.3 m, and *Rhizophora* seedlings taller than 50 cm but shorter than 1.5 m; Class III = *Laguncularia* and *Avicennia* saplings with a diameter <2.5 cm and a height >1.3 m and *Rhizophora* saplings >1.5 m (Valdez-Hernández 2002a). In the field, it was noted if saplings originated from coppicing or from seeds.

3.5.2. Tree Diameter Distribution Analysis

Four different analyses and charts were conducted and used to evaluate forest structure by diameter size classes. One analysis considered diameter distributions by 1 cm diameter size classes and tree height by mangrove species. For this model a Kernel Density Estimator was used, creating a histogram chart for each species at each forest condition. The Kernel Density Estimator was displayed over the observed data as a curved line, which consists in overlapping a probable frequency of trees within a diameter size class with each field observations recorded (Seaman and Powell 1996). A frequency estimate was obtained for each point considering the total in the sample, which results in an averaged frequency for each diameter size class. The second analysis evaluated *Laguncularia* diameter distribution analysis by 2 cm diameter size classes to facilitate comparisons with similar studies in the region. The third analysis employed a single Weibull analysis to determine the incidence of young trees grouped in 5 cm diameter classes. A fourth analysis employed a bivariate model. The minimum tree diameter used in all the models was 2.5 cm.

3.5.3. Mangrove Standing Wood Volume

Standing wood volume was calculated using tree diameter (cm) and height (m) measurements according to the site-specific formula generated for *Laguncularia* in the San Blas Mangrove System (Valdez-Hernández 2004).

3.6. Historical Mangrove Cover Change Analysis

I used the Mexican Mangrove National Distribution Map 2010, scale: 1:50,000 (CONABIO 2013c), and the GIS map of the Management Unit for Wildlife Conservation (UMA) of the Ejido San Blas 2010 to determine the mangrove area within the Management Unit. I also used the

Mexican Mangrove National Distribution Maps (CONABIO 2013a, b, and c) for the periods 1970/1980, 2005 and 2010 to determine if the mangrove cover in the Management Unit increased or decreased during these periods. I overlapped CONABIO's maps with the map of the Management Unit of the Ejido San Blas to estimate the mangrove cover area for each period. As the surface data contained on CONABIO's maps did not coincide with the surface of the Management Unit, mangrove cover areas were recalculated and adjusted to the UMA surface using ArcMap from ArcGIS 10.2.2.

3.7. Statistical and Data Analysis

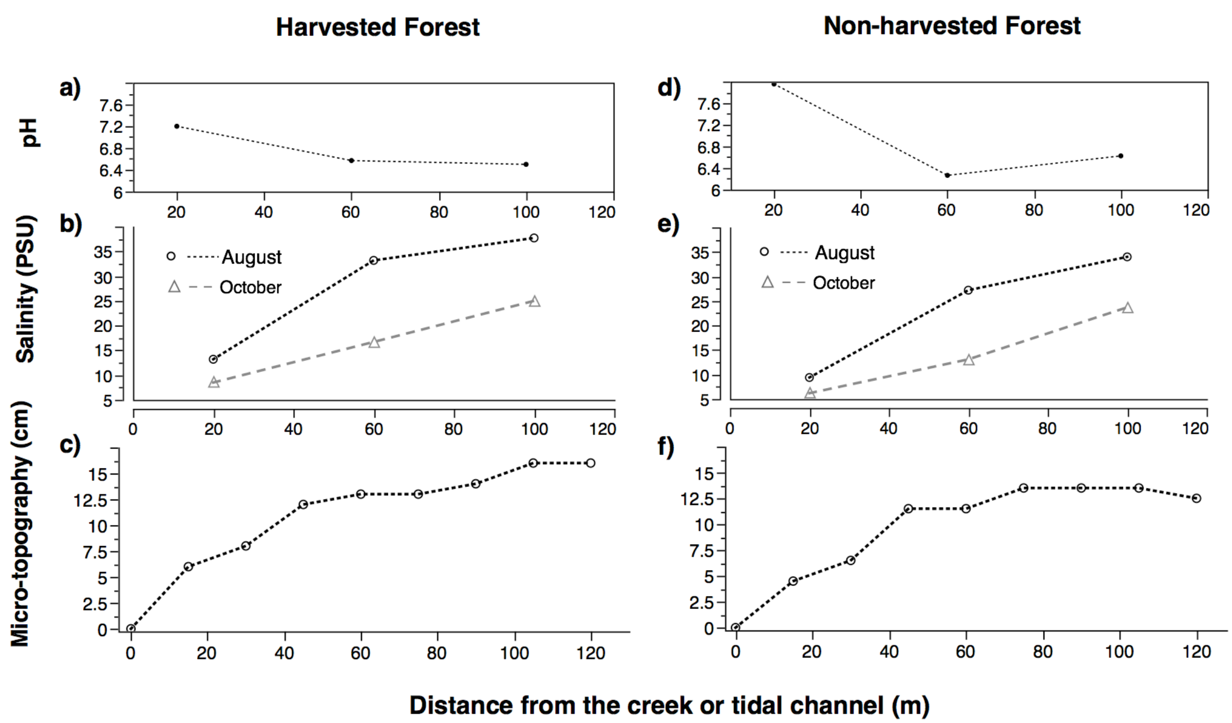
JMP® 11.2.0 from SAS® was used to perform statistical analyses and charts. PhythonTM was only used to perform diameter distribution statistical analyses and create charts. Data was analyzed statistically with a randomized block design. The three inundation zones parallel to the edges of the creeks were used as blocks, and the two conditions of mangrove forest management, harvested and non-harvested, were used as treatments. An alpha of 0.05 was used to test data for normality using a Shapiro-Wilk test. To test for differences in tree density, canopy height, DBH, canopy cover, basal area, wood volume, and natural regeneration between harvested and non-harvested forests stands, one-way analysis of variance (ANOVA) was used for normally distributed data. When the analysis of variance showed significant differences, a multiple comparison Tukey-Kramer (α =0.05) test was used to determine which means were different. For non-normally distributed data, a nonparametric Wilcoxon Test was used. Regression analysis was used to determine the relationship between various variables. Only probability values $p<0.05$ were considered significantly different. Results are presented as averages ± one standard deviation. All maps included in this thesis were created using ArcMap from ArcGIS 10.2.2.

Chapter 4. Results

4.1. Physicochemical and Micro-elevation Gradients

Harvested and non-harvested stands showed similar environmental conditions. Salinity increased from low to high tidal zones ranging from 6 to 38 PSU. The elevation gradient explained 63% of the variability ($p > 0.0013$) (Figure 6b,e). Micro-topography increased by 16 cm in the harvested stand and by 13.5 cm in the non-harvested stand (R_{adj}=0.783, $p > 0.0001$) (Figure 6c,f).

Figure 6. Physicochemical and micro-elevation gradients. pH, salinity and micro-topography in harvested (a, b, c) and non-harvested forests (d, e, f).



4.2. Mangrove Forest Structure

Species composition varied between harvested and non-harvested stands. *Rhizophora* was present in both sites while *Avicennia* was found only in harvested forests. According to the % of species relative dominance, in harvested forests, two combinations of mangrove species were found in subplots of 100 m^2: a) *Laguncularia-Avicennia-Rhizophora*, and b) *Laguncularia-*

Avicennia. Four subplots out of the nine in harvested forests contained a *Laguncularia-Avicennia-Rhizophora* assemblage, two with more *Avicennia* than *Rhizophora* trees, and two with an equal number of *Avicennia* and *Rhizophora* trees. The other five plots contained a *Laguncularia-Rhizophora* assemblage. Non-harvested subplots contained either: a) *Laguncularia-Rhizophora*, or b) *Laguncularia*. Seven of them contained a *Laguncularia-Rhizophora* assemblage and two only *Laguncularia*.

Tables 4, 5 and 6 report forest structure of harvested and non-harvested stands. Overall, analysis of variance (ANOVA) revealed no significant differences in average tree densities between harvested and non-harvested forests considering either all species recorded or only *Laguncularia* (see Table 4 and 5). Although the average density of *Laguncularia* trees in harvested stands was 1.7% higher than in non-harvested stands, the difference was not significant (Table 5). Average tree density of non-*Laguncularia* species was similar in both forest stands with 378 and 400 ha^{-1} in harvested forests and non-harvested, respectively. Tree densities among tidal zones within sites were not significantly different in either harvested or non-harvested forests when considering only *Laguncularia* individuals or all species (Table 5).

Average tree height in non-harvested stands was significantly higher than in harvested stands (13.2 ±2.5, 10.7 ±1.9 m, respectively). Tree height differed significantly among tidal zones only in harvested stands with higher canopy height in low tidal zones than mid and high zones. Tree diameter at breast height (DBH) was not significantly different between harvested and non-harvested stands (8.4 ±4.5 and 12.0 ±2.5, respectively). Canopy cover ranged from 53% to 68% within 100 m^2 subplots, with an ecosystem average of 61%. Canopy cover was not statistically significant within tidal zones in either harvested or non-harvested forests. However, harvested stands had higher canopy cover than non-harvested forests (66 ±0.07% and 57

±0.09%, respectively). Average basal area of *Laguncularia* was significantly different between sites, with higher basal area in non-harvested stands (Table 6). Nevertheless, within each site, differences in the basal area were not detected along the tidal gradient.

Table 4. Mangrove forest structure in harvested and non-harvested stands by species in the San Blas Mangrove System. DBH = diameter at breast height.

	Laguncularia racemosa		***Avicennia germinans***	***Rhizophora mangle***	
Forest attributes	**Non-harvested**	**Harvested**	**Harvested**	**Non-harvested**	**Harvested**
Tree density (ind. ha^{-1})	1,978	3,389	378	311	89
Relative dominance (%)	97	86	12	3	1
Basal area (m^{-2} ha^{-1})	29	17	2	1	0.28
Tree height (m) ranges	3-22	5-21	4-18	2-16	7-14
DBH (cm) ranges	4-34	3-34	6-30	3-12	5-8

Table 5. Tree density by tidal zone in harvested and non-harvested *Laguncularia* dominated stands.

Mangrove Condition	**Tidal Zone**	***Laguncularia* Density (trees ha^{-1})**	**All species Density (trees ha^{-1})**
Harvested	Low	3,400	4,100
	Mid	3,233	3,433
	High	3,533	4,033
	Average ±sd	3,389 ±150	3,855 ±367
Non-harvested	Low	2,033	2,300
	Mid	1,567	2,033
	High	2,333	2,533
	Average±sd	1,978 ±386	2,289 ±250

Table 6. Basal area by tidal zone in harvested and non-harvested stands.

Mangrove Condition	**Tidal Zone**	***Laguncularia* Basal Area (m^2 ha^{-1})**	**All species Basal Area (m^2 ha^{-1})**
Harvested forest	Low	20.2	22.8
	Mid	14.3	14.9
	High	16.8	22.7
	Average±sd	17.1 ±3.0	20.1 ±4.5
Non-harvested forest	Low	30.3	30.7
	Mid	26.3	27.3
	High	29.5	30.5
	Average±sd	28.7 ±2.1	29.5 ±1.9

4.3. Natural Regeneration

Although natural regeneration originating from both seeds and vegetative propagation was observed for *Laguncularia*, *Rhizophora* and *Avicennia* in both harvested and non-harvested forests, only natural regeneration originating from seeds was recorded for the three species within the subplots evaluated (Figures 7, 8 and 9). High levels of vegetative propagation of *Laguncularia* through coppicing was observed outside subplots in the harvested forest, assumed to be due to vegetative propagation enhanced as the result of harvesting.

Seedlings (Class I and Class II) were recorded in both harvested and non-harvested stands with differences found among tidal zones and mangrove species. Saplings (Class III) were only recorded in non-harvested stands, despite high densities observed outside the evaluated subplots in harvested stands. Overall, ANOVA analysis showed no significant difference in seedlings (Class I and Class II) within each tidal zone between harvested and non-harvested stands for the three mangrove species (Table 7). However, numbers of seedlings and saplings among tidal zones within sites were significantly different (Table 7, Figure 10). In both stands, higher seedling densities were recorded in low tidal zones and lower densities were recorded in high tidal zones (Table 7). Differences were also recorded in the number of seedlings per height class (Table 8, Figure 10). Seedlings in Class I showed no significant differences between harvested and non-harvested stands; high variability was observed within Class II, and Class III was only present in non-harvested stands. Overall, only Class I and II were present in harvested stands and found in all the three tidal zones better represented by *Laguncularia* individuals.

ANOVA analysis showed differences in natural regeneration among mangrove species (Table 8, Figure 10). Seedlings from *Laguncularia* (Class I and Class II) were recorded in the three tidal zones in harvested stands while in non-harvested stands seedlings from *Laguncularia*

Class I were only present in mid tidal zones, seedlings from Class II in low tidal zones, and in high tidal zones seedlings were not present. Samplings (Class III) of *Laguncularia* were only recorded in low tidal zones in the non-harvested stand. *Laguncularia* seedlings from Class I and II were the dominant natural regeneration recorded in harvested forests. *Rhizophora* natural regeneration from seeds was present only in non-harvested forests stands. *Rhizophora* seedlings Class II and saplings were recorded within the three tidal zones in non-harvested stand while seedlings Class I were only recorded in mid tidal zones. Only seedlings from *Avicennia* in Class II were observed in low and mid tidal zones in harvested stands (Table 8, Figure 10).

Table 7. Natural regeneration per tidal inundation zones in harvested and non-harvested stands.

Mangrove Condition	Tidal Zone	Seedlings (ind. ha^{-1})	Saplings (ind. ha^{-1})
Harvested forest	Low	13,866 ±14,589	0
	Mid	11,500 ±5,972	0
	High	2,000 ±1,697	0
	Average±sd	10,178 ±9,468	0
Non-harvested forest	Low	20,000 ±14,142	5,600 ±6,223
	Mid	14,000 ±8944	10,000
	High	5,400 ±6,505	800
	Average±sd	13,422 ±9,869	6,400 ±4,400

Table 8. Natural regeneration per class by mangrove species in the San Blas Mangrove System.

Mangrove Condition	Mangrove species	Seedlings (ind. ha^{-1})		Saplings (ind. ha^{-1})
		Class I	Class II	Class III
Harvested forest	*L. racemosa*	16,933 ±14,840	6,400 ±3,418	0
	A. germinans	0	7,200 ±4850	0
Non-harvested forest	*L. racemosa*	10,000 ± 0	10,000	10,000
	R. mangle	10,000	16,160 ±13,510	5,500 ±5199

Note: In non-harvested forest Class II and III *L. racemosa* was recorded in only one subplot, thus there is no standard deviation.

Figure 7. Recently harvested *Laguncularia* trees with resprouts (circled).

Figure 8. Standing wood of *Laguncularia* in harvested stands: a) stems originated from seeds, and b) stems originated by coppicing.

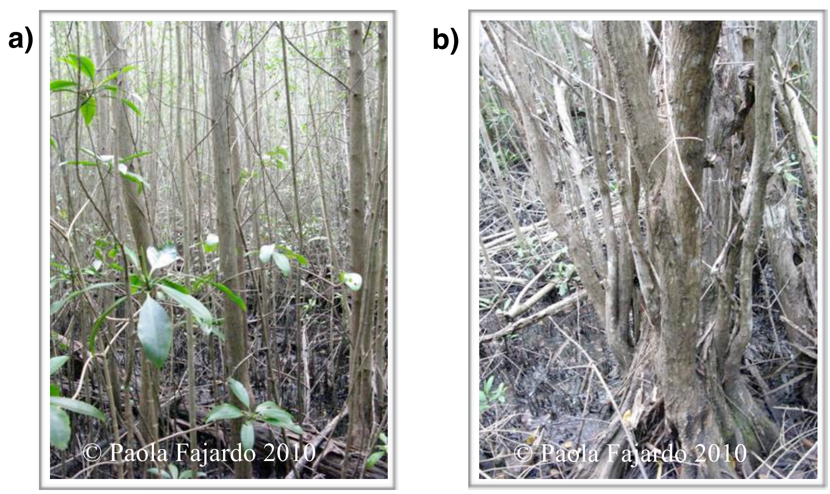


Figure 9. Natural regeneration in the San Blas Mangrove System: a) natural regeneration outside subplots and b) subplots without natural regeneration in harvested stands.

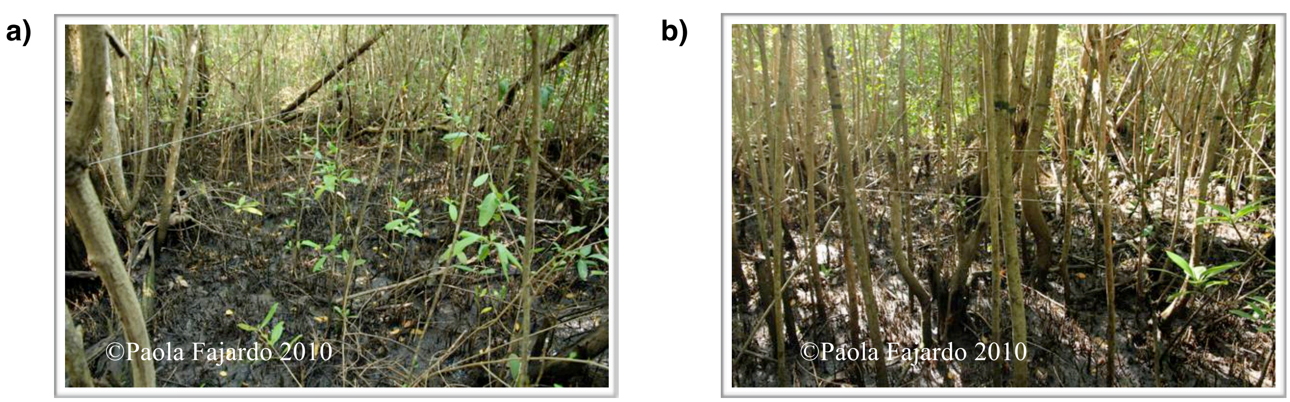
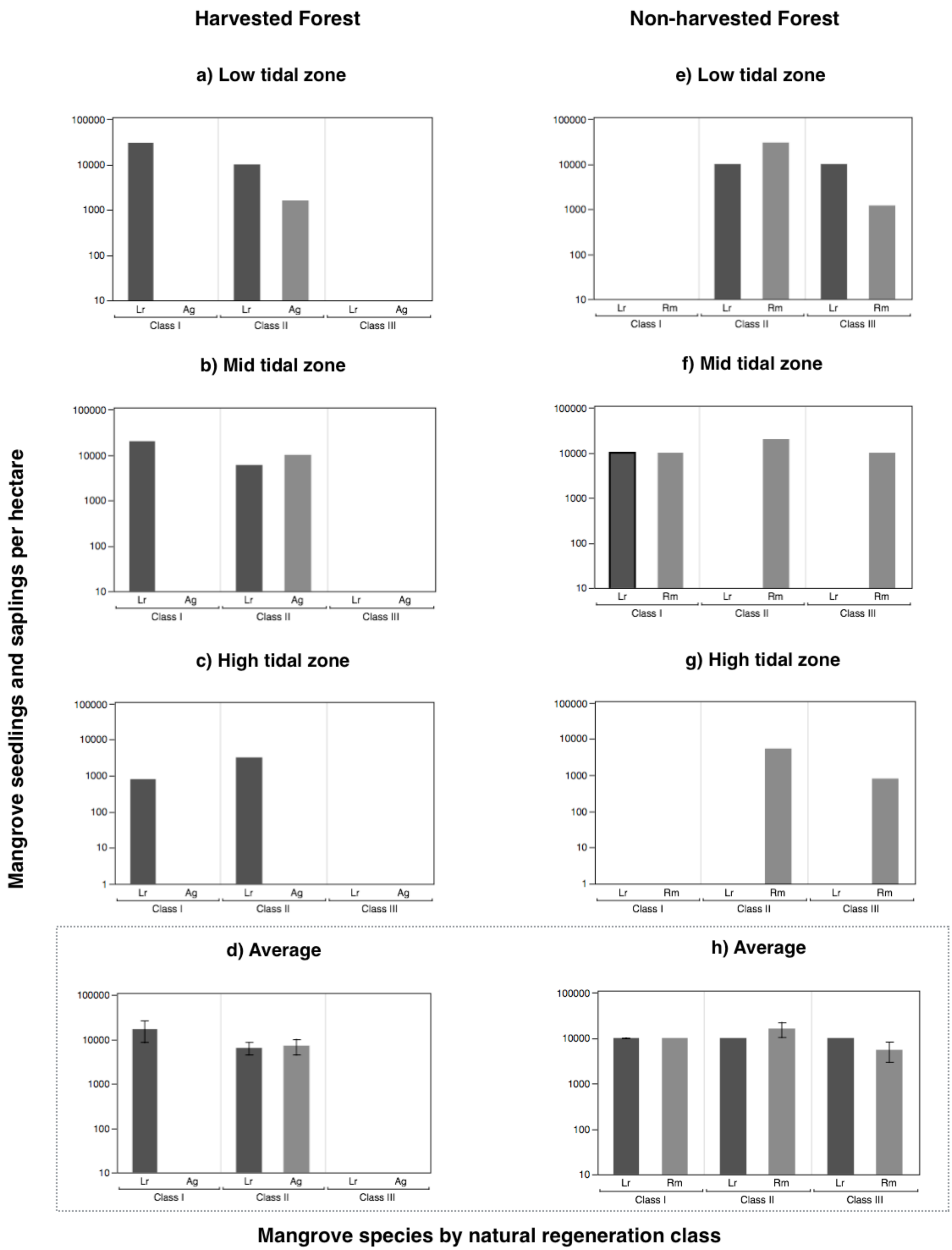


Figure 10. Natural regeneration by height class within tidal zones in harvested (a-d) and non-harvested mangrove stands (e-h).



4.4. Tree Diameter Distribution Analysis

Different analyses were conducted to evaluate forest structure by diameter size class: 1) Diameter distribution by 1 cm size classes and tree height by mangrove species, 2) *Laguncularia* diameter distribution analysis by 2 cm size classes, 3) a single Weibull analysis to determine the incidence of young trees grouped in 5 cm diameter size classes, and 3) a bivariate model with young trees also grouped in 5 cm diameter size classes.

4.4.1. Tree Diameter Distribution Analysis by 1 cm Size Classes and Tree Height

Figures 12, 13, 14 and 15 show the diameter distribution of 1 cm diameter size classes, as well as the correspondent tree height in harvested and non-harvested stands.

Avicennia trees were only recorded in plots of harvested stands, although individuals of *Avicennia* were also observed near plots in non-harvested stands. Diameter analysis of *Avicennia* trees showed the highest diameter frequency recorded in trees with a DBH of 5 cm, followed by trees with a DBH of 8 cm. Small to medium trees were represented by individuals with diameters that ranged from 2.5 to 10 cm. However, some tall trees with small diameters were also recorded. While the majority of tall trees were represented by individuals with diameters that ranged from 12 to 30 cm (Figure 12).

Rhizophora trees were recorded in both harvested and non-harvested stands, with higher densities found in non-harvested stands. *Rhizophora* diameter distribution analysis showed that only trees with small DBHs between 2.5 and 7 cm were recorded in harvested stands, but which were medium to tall in height (Figure 13a). In non-harvested stands the dominant *Rhizophora* trees were medium to tall trees with DBHs between 4 and 6 cm (Figure 13b). Medium to tall *Rhizophora* trees with DBHs between 8 and 12 cm were also recorded with low densities.

As *Laguncularia* is the dominant species in the stands evaluated, a major diversity in diameter size classes was observed. In harvested stands, there was a higher frequency of *Laguncularia* trees with smaller diameters and a normal distribution of trees with diameters from 2.5 to 15 cm (Figure 14a). High diameter frequencies of *Laguncularia* were observed in trees with DBHs from 2.5 to 10 cm, with the highest reported for DBHs between 5 and 7 cm (Figure 11). However, a more homogeneous diameter distribution of *Laguncularia* trees was found in non-harvested forests (Figure 14b). Results showed that trees of *Laguncularia* with small DBH could have the same height as trees with large DBH. It was observed that a single DBH size class could occur with multiple tree heights (Figure 12, 13 and 14). Diameter frequency (trees ha^{-1}) of the three mangrove species in harvested stands, non-harvested stands and at the ecosystem level are shown in Figure 15. Overall, analysis of diameter distribution showed that the stands evaluated correspond to uneven-aged forests in both harvested and non-harvested stands. Stands are considered uneven-aged following local villagers information regarding stand age and harvesting rotation schemes, as well as due to the variety of tree DBHs size classes registered.

4.4.2. *Laguncularia* Tree Diameter Distribution Analysis by 2 cm Size Classes

Further diameter distribution analyses by 2 cm size classes were performed only for *Laguncularia* to be able to facilitate the comparisons with previous studies.

To evaluate the sustainability of harvesting practices *Laguncularia* trees with a DBH >2.5 were grouped in 2 cm classes producing 18 classes from 2.5 to 38 cm. No individuals were recorded in the 26 cm class for either forest condition. In harvested stands, 14 diameter size classes were present while 16 diameter size classes were recorded in non-harvested stands. Harvested forests and non-harvested forests shared 12 diameter categories. Each DBH size class was compared between harvested and non-harvested stands to determine differences in tree density (Figure 16), basal area (Figure17), and commercial standing wood volume (Figure 19). The results corresponding to wood volume are presented in a separate section. Overall, only small differences in tree density and basal area were observed among size classes shared between forests conditions. Tree density significantly varied in only two DBH size classes; Class 6 had higher tree densities in harvested stands and Class 12 in non-harvested stands (Figure 16). Basal area of trees in DBH Classes 4 and 6 was greater in harvested stands while basal area was higher in class 12 in non-harvested stands (Figure 17).

4.4.3. Weibull Distribution and Bivariate Analysis of *Laguncularia* Trees

The common Weibull distribution was fitted to *Laguncularia* trees grouped in 5 cm DBH classes, which resulted in 7 diameter size classes in harvested stands, and 8 diameter size classes in non-harvested stands (Figure 18). The Weibull analysis resulted in a clear reverse J-shaped or negative exponential curve shape in harvested stands (Figure 18a), and a less clear reverse J-shaped line in non-harvested stands (Figure 18b). Several studies have modeled tree diameter distribution in order to improve forestry practices using the Weibull function (e.g. Bailey and Dell 1973; Maltamo *et al.* 2000), including for modeling the rotated-sigmoid or J inverse shape curves in uneven-aged forest stands (e.g. Zhang *et al.* 2001). If a diameter distribution analysis results in a reverse J-shaped curve form, it is considered that balanced, uneven-aged diameter distributions are occurring in the evaluated stand. It has also been suggested that a reverse J-shaped curve represents an all-aged stand or forest (Wittwer *et al.* 1990). This analysis showed, that the frequency of trees in the 0-5 and 5-10 cm DBH classes was 3.6 and 2.2 times, respectively, higher in harvested stands.

A bivariate analysis showed strong linear relationships between the number of individuals and the DBH size class. Small DBH size classes had a higher frequency of individuals. A stronger relationship occurs in non-harvested stands ($R^2_{adj} = 0.852$, $\alpha=0.05$), where a higher number of trees were recorded in some medium to large DBH trees. However, in harvested stands, there was double the number of individuals in small DBH classes compared with non-harvested stands. It is important to highlight that despite harvesting activities DBH size classes from small to large were well represented in harvested stands.

Figure 11. *Laguncularia* stem density of small DBH size classes in harvested-stands.

Figure 12. Diameter frequency (trees ha^{-1}) of *Avicennia* in harvested stands. Brown vertical lines represent tree height (m) and the size of filled green circles at the top of the line are proportional to the DBH (cm) of each tree. The histogram represents the empirical diameter distribution of 5 cm DBH size classes.

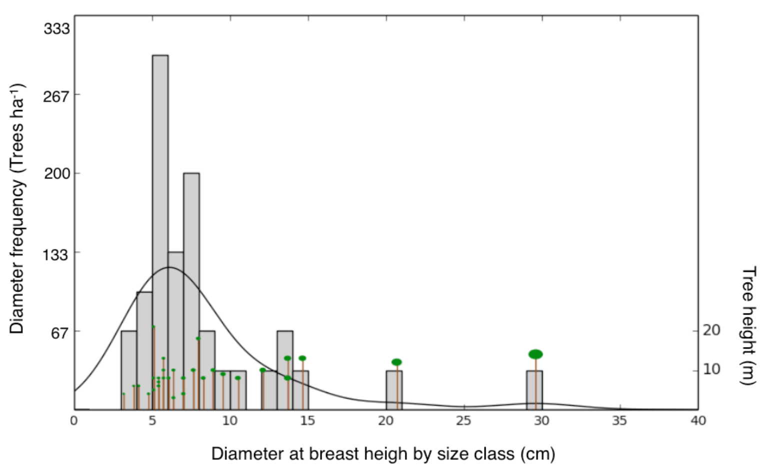


Figure 13. Diameter frequency (trees ha^{-1}) of *Rhizophora* in a) harvested stands, b) non-harvested stands and c) all sites. Brown vertical lines represent tree height (m) and the size of filled green circles at the top of the line are proportional to the DBH (cm) of each tree. The histogram represents the empirical diameter distribution of 5 cm DBH size classes. Kernel density estimator = curve line.

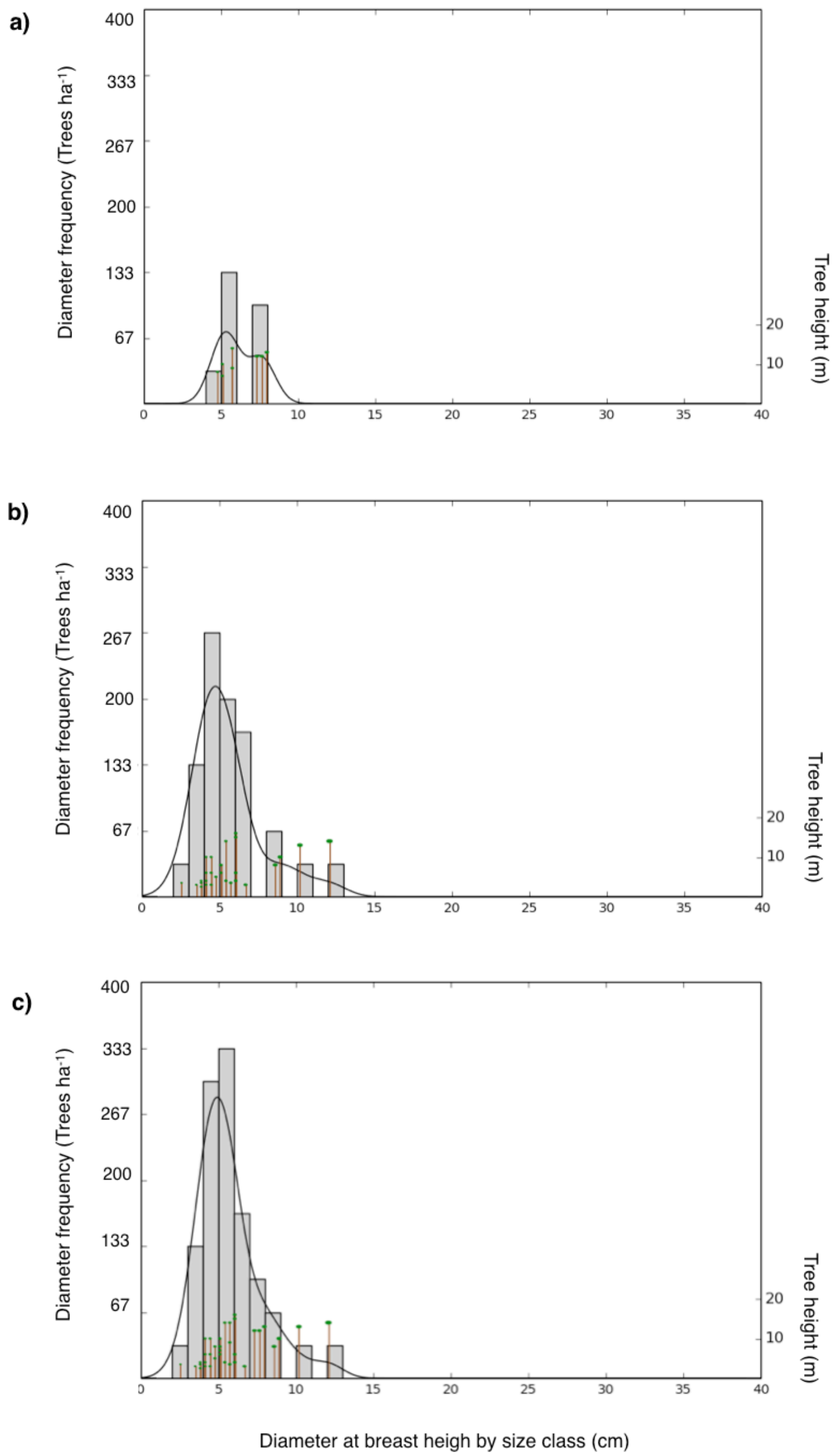


Figure 14. Diameter frequency (trees ha^{-1}) of *Laguncularia* in a) harvested stands, b) non-harvested stands and c) all sites. Brown vertical lines represent tree height (m) and the size of filled green circles at the top of the line are proportional to the DBH (cm) of each tree. The histogram represents the empirical diameter distribution of 5 cm DBH size classes.

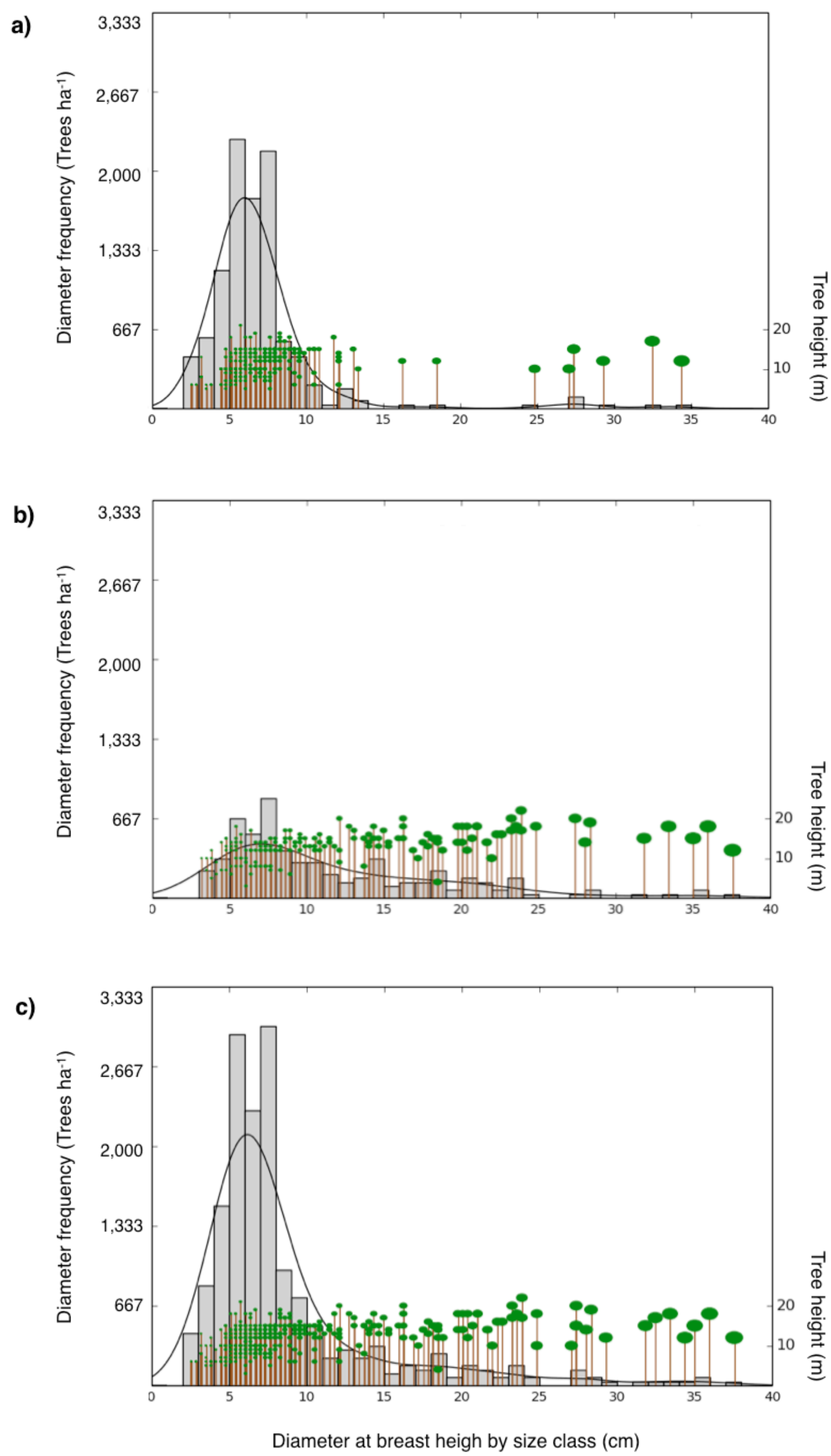


Figure 15. Diameter frequency (trees ha^{-1}) of the three mangrove species in a) harvested stands, b) non-harvested stands and c) all sites. Trees in the graph are indicating tree height (m). The size of the canopy is proportional to the DBH of each tree.

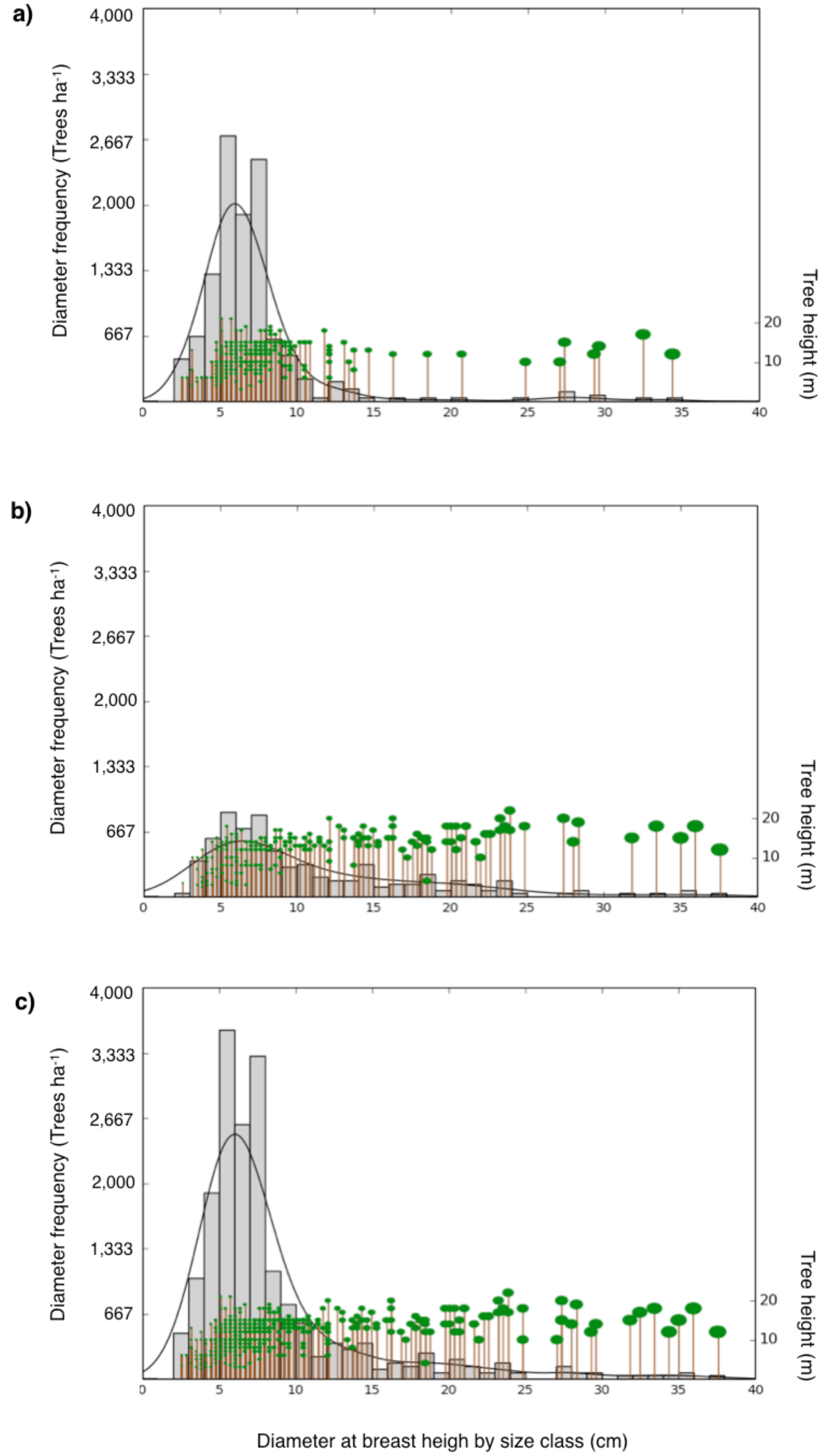


Figure 16. ***Laguncularia* density (trees ha^{-1}) by diameter at breast height by size class (cm) within tidal zones and total averages per site in harvested (a-d) and non-harvested forest stands (e-h).**

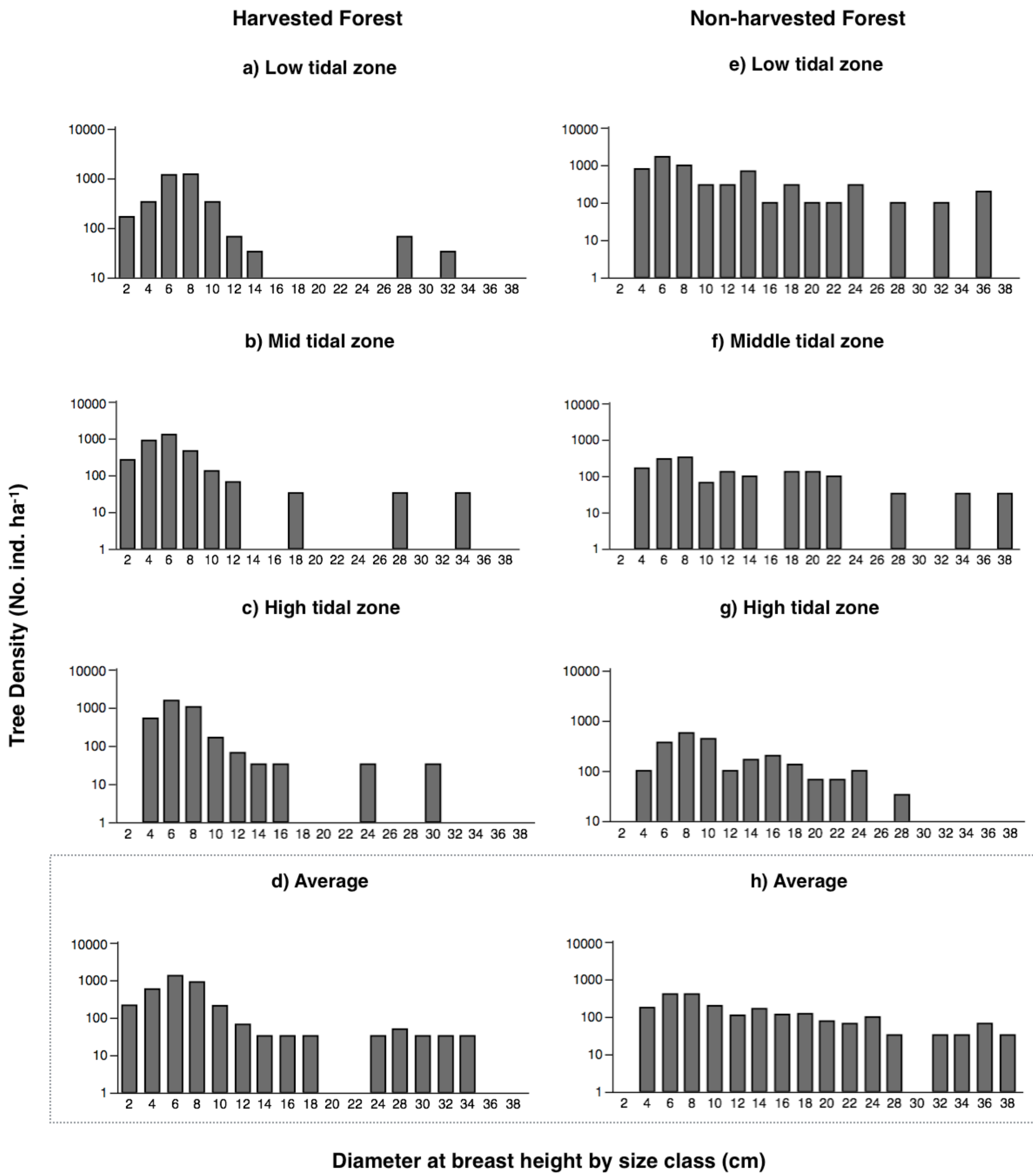


Figure 17. ***Laguncularia*** **basal area (m^2 ha^{-1}) by DBH size class (cm) within tidal zones and total averages in harvested (a-d) and non-harvested stands (e-h).**

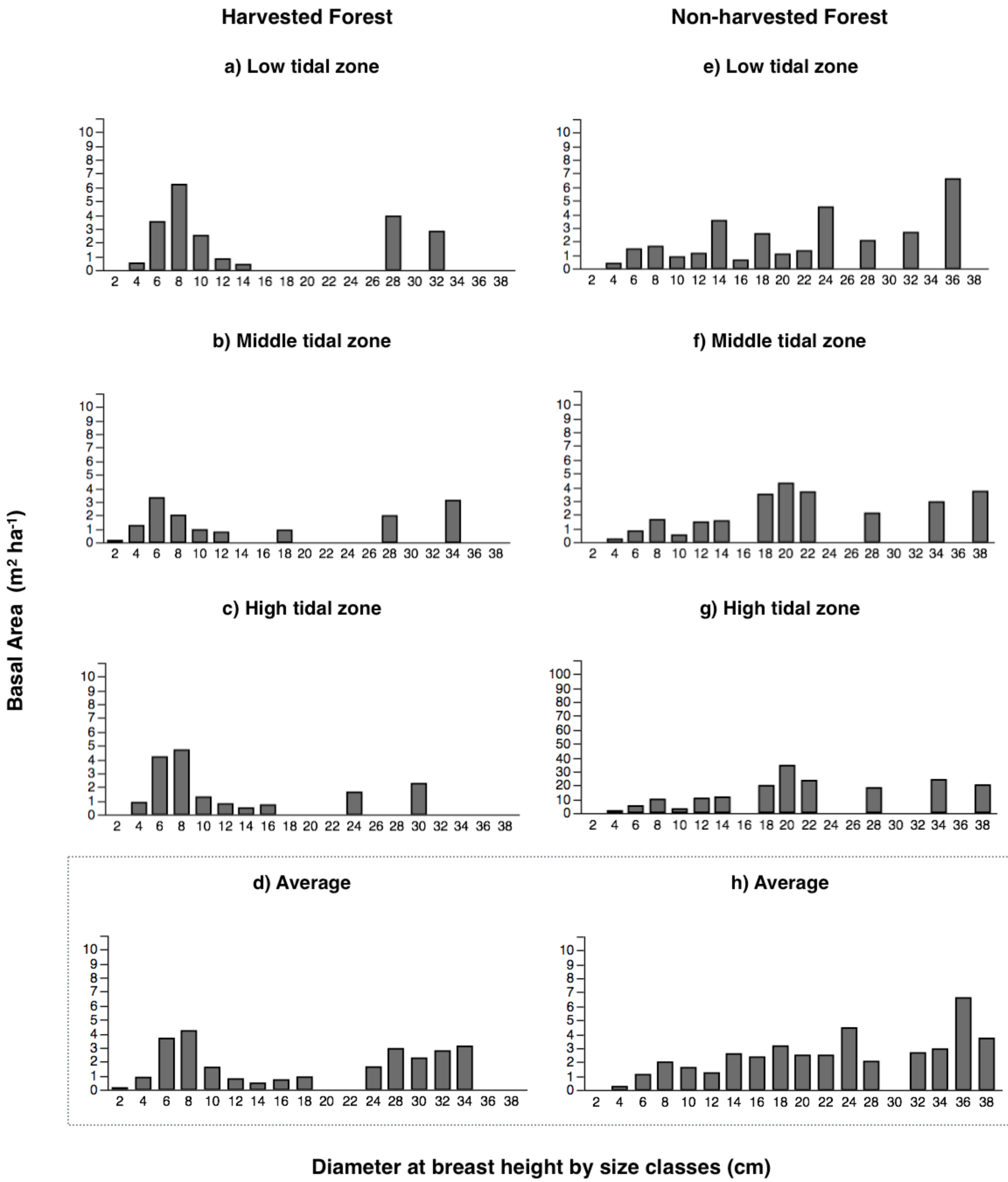
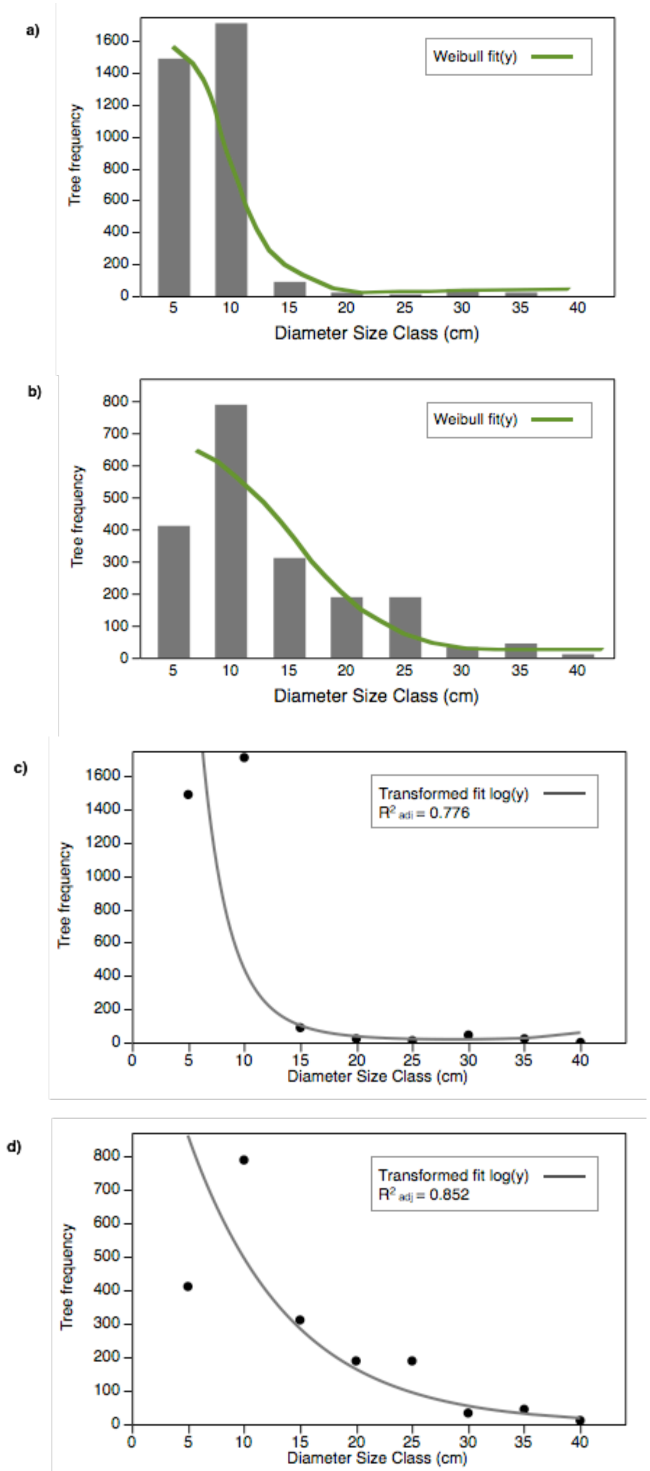


Figure 18. Tree frequency distribution per hectare by 5 cm DBH size classes of *Laguncularia* in both harvested and non-harvested stands. Weibull analysis (α=0.05) indicates the shape of the tree frequency distribution (y) by DBH size class (x) in a) harvested, and b) non-harvested stands. Bivariate analysis "log(y) regression" (α=0.05) indicates the relation between tree frequency (y) by DBH size class (x) in c) harvested, and d) non-harvested stands.



4.5. Mangrove Standing Wood Volume

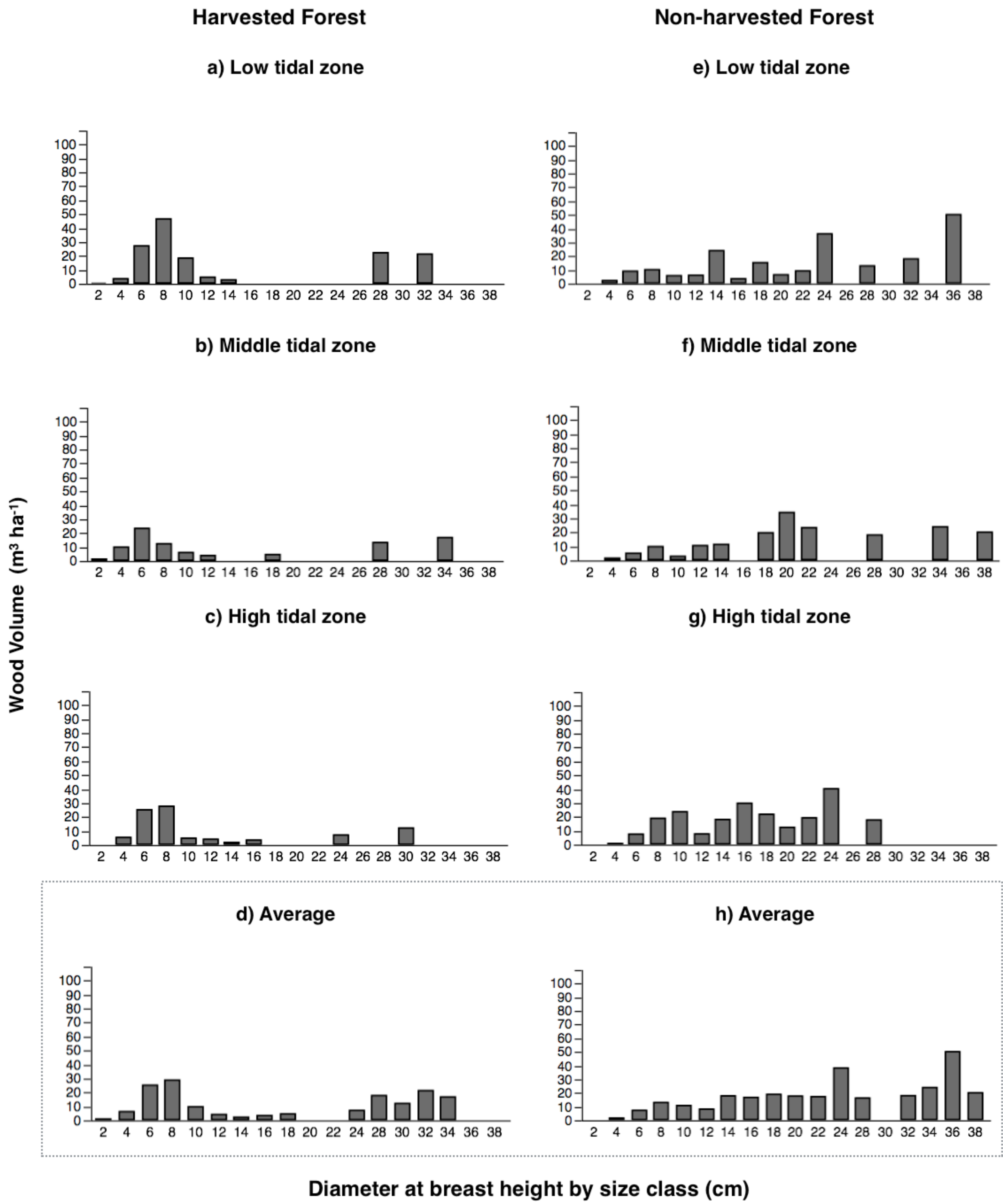
Overall, results showed large volumes of standing wood in the evaluated stands compared with other mangroves elsewhere. Within each forest condition, there were no significant differences in wood volume by tidal zones. Interestingly, low tidal zones (20 to 30 m inland) in harvested stands had slightly higher wood volumes considering either just *Laguncularia* or all species. These are transition areas between the protected zones (up to 10-25 m inland) for water body protection where harvesting is not allowed. However, ANOVA analysis showed that the average *Laguncularia* average standing wood volume within the 300 m^2 evaluated plots significantly differed between harvested and non-harvested stands (Table 9). The average wood volume of non-*Laguncularia* (*Avicennia* plus *Rhizophora*) species in non-harvested stands was 18.43 m^3 ha^{-1}, more than twice the wood in harvested stands (5.23 m^3 ha^{-1}).

In harvested forests, *Laguncularia* wood volume by diameter size class revealed higher volume within the 6 and 8 cm DBH size classes (Figure 19). Diameter analysis also showed high wood volumes in large DBH size classes from 28 to 34 cm. In non-harvested stands, the DBH size classes 24 and 36 showed the highest wood volumes of *Laguncularia.* In contrast to harvested stands, the lowest wood volumes in non-harvested stands corresponded to small DBH size classes from 4 to 12 cm, and medium volume estimates for the rest of the DBH size classes.

Table 9. Average standing wood volume (m^3 ha^{-1}) by tidal zone in harvested and non-harvested 300 m^2 *Laguncularia* dominated stands.

Mangrove Condition	Tidal Zone	*Laguncularia*	All species
		Wood Volume (m^3 ha^{-1})	
Harvested forest	Low	151.5	167.2
	Mid	96.2	100.2
	High	96.3	132.0
	Average	114.7 ±31.0	133.1 ±33.5
Non-harvested forest	Low	216.6	219.1
	Mid	187.0	193.7
	High	224.4	230.8
	Average	209.3 ±19.7	214.5 ±19.0

Figure 19. ***Laguncularia*** **wood volume (m^3 ha^{-1}) by DBH size class (cm) within tidal zones and total averages in harvested (a-d) and non-harvested stands (e-h).**



4.6. Historical Mangrove Cover Change Analysis

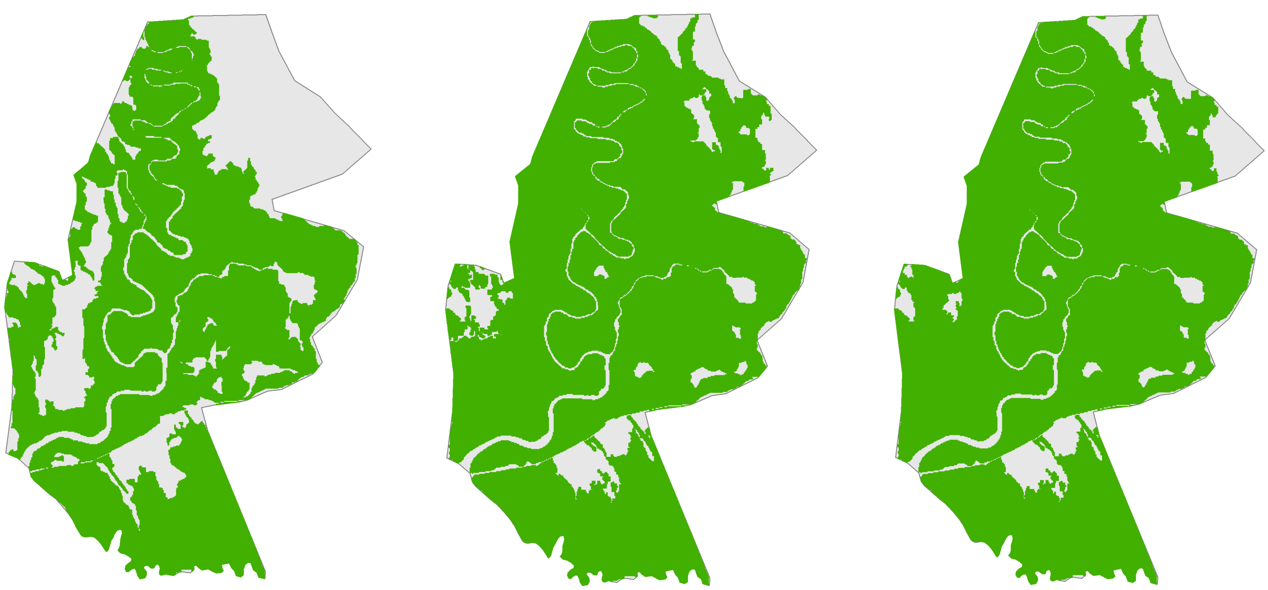
Historical mangrove cover maps from the periods 1970/1980, 2005 and 2010 created by the Mexican National Commission for the Knowledge and Use of Biodiversity (CONABIO, acronym in Spanish), revealed mangrove expansion within the Management Unit for Wildlife Conservation (UMA, acronym in Spanish) located in the San Blas Mangrove System (Figures 20 and 21). Mangrove cover estimated for each period was divided into four categories: 1) no changes, 2) gains, 3) losses, and 4) net gain (the difference between gains and losses) as shown in Table 10. Historical mangrove cover analyses using these four categories are critical for evaluating the sustainability of wood production, with mangrove cover changes monitoring at least every five years, as to detect any possible degradation of the system caused by the harvesting. Mangrove forests in the Management Unit expanded in the last 40 years, with the highest rates between the 1970/1980 and 2005, with a net gain of 280 ha (Figure 20). From 1970/1980 to 2005 total mangrove cover within the Management Unit increased 273%, and from 2005 to 2010 the increase was 1% ha (Figure 20). Mangrove vegetation expanded in coastal plains, mainly due to the replacement of water bodies and other wetlands, and to a lesser degree, due to replacement of wetlands in the southern part of the Management Unit (Figure 21). Results of the historical mangrove cover analysis in the Management Unit per zone are shown in Table 11. The major increase in mangrove cover observed in the Management Unit was from 1970/1980 to 2005 (32%) in conservation areas, followed by an increase of 15% in the Production Zone #2. Production Zone #1 was the only area that had a reduction of mangrove cover from 1970/1980 to 2005, but the decrease was only 3% in 35 years. An increase of mangrove cover occurred in Production Zones #2 and #3 with 16% and 8% gains, respectively over 35 years. From 2005 to 2010 mangrove cover did not change in the three production zones.

Table 10. Historical mangrove cover changes in the UMA of the Ejido San Blas, Nayarit.

Period (year)	Historical Mangrove Cover Change at the UMA level	Surface area (ha)
1970/1980-2005	No change	996.14
	Gain	324.33
	Loss	44.16
	Net gain	280.17
2005-2010	No change	1,316.24
	Gain	17.87
	Loss	4.23
	Net gain	13.64

Figure 20. Historical mangrove cover changes in the UMA of the Ejido San Blas for the periods a) 1970/1980, b) 2005 and c) 2010.

a) Mangrove cover 1970/1980 **b) Mangrove cover 2005** **c) Mangrove cover 2010**



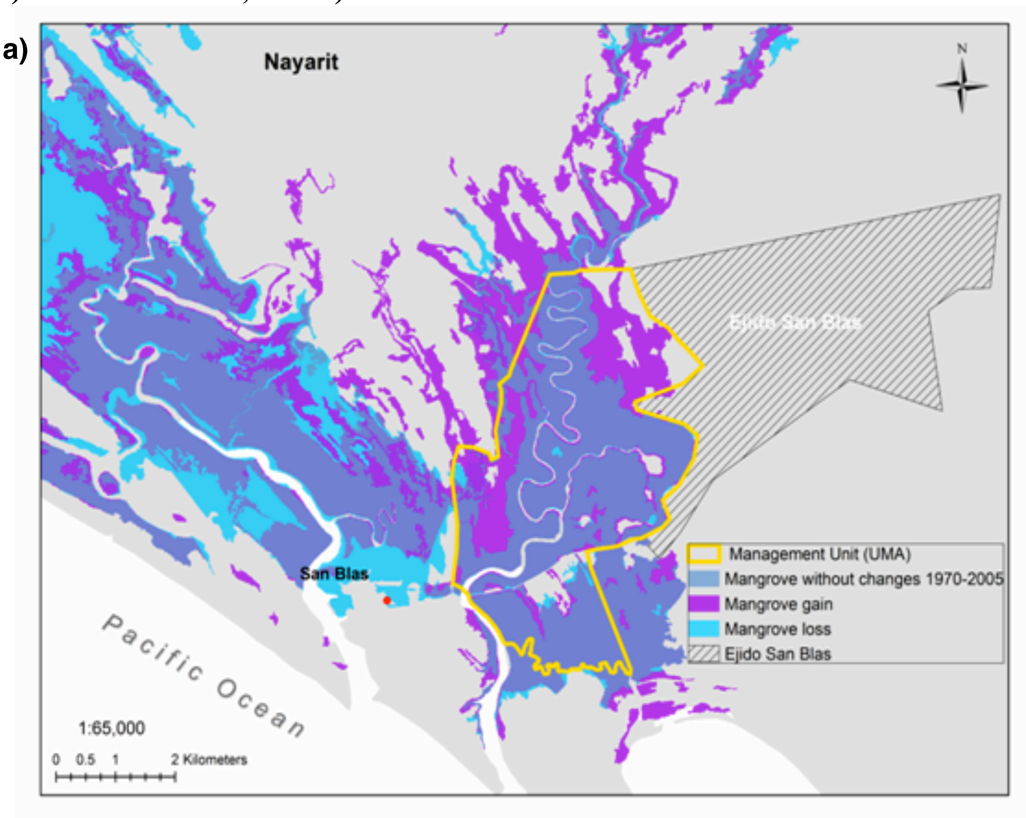
1,040 hectares **1,320 hectares** **1,334 hectares**

Table 11. Historical mangrove cover (ha) in the UMA of the Ejido San Blas per zone.

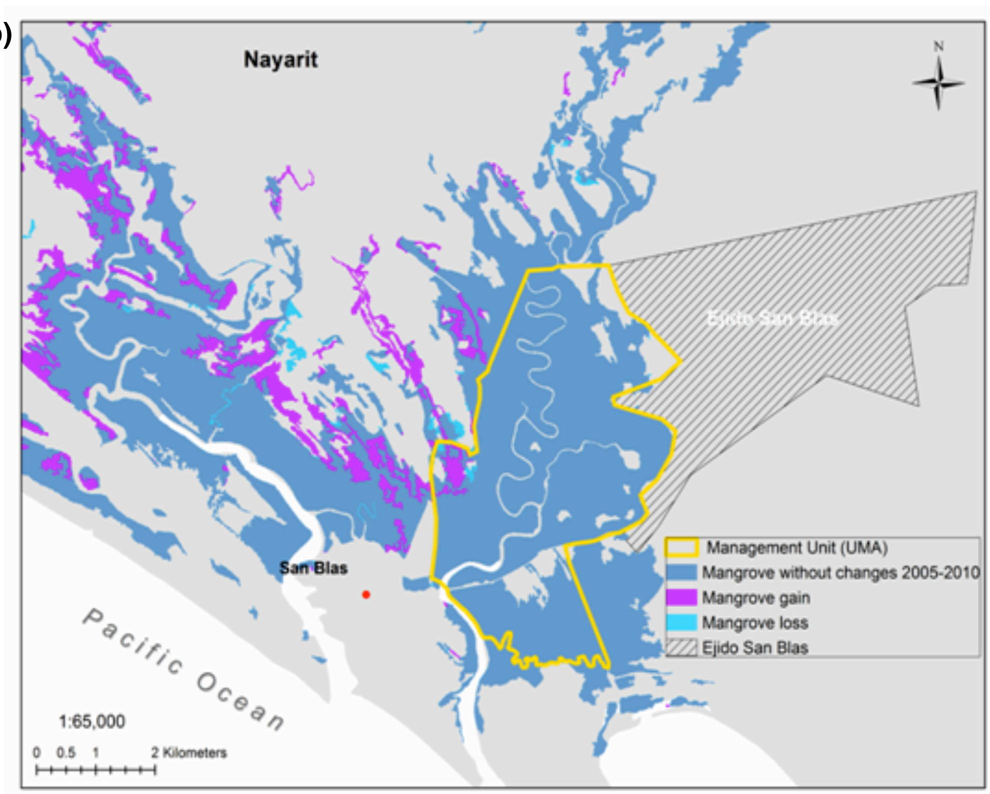
Year	Total surface (ha)	Protection zones (ha)	Conservation zones (ha)	Production Zone #1 (ha)	Production Zone #2 (ha)	Production Zone #3 (ha)
1970/1980	1,040.30	54.49	778.12	61.21	88.81	57.67
2005	1,320.46	65.97	1,029.60	59.65	102.74	62.5
2010	1,334.11	65.97	1,043.25	59.65	102.74	62.5

Figure 21. Historical mangrove cover changes in the UMA of the Ejido San Blas for the periods a) 1970/1980-2005, and b) 2005-2010.

a)



b)



Chapter 5. Discussion

5.1. Evaluating the Sustainability of Community-based Selective Small-scale Harvesting in the Mangrove Management Unit of the Ejido San Blas

Mangrove forests provide multiple ecosystem services and cover only a subtle global surface between 0.34 to 0.38% (estimated from Spalding *et al.* 2010; Giri *et al.* 2011 data) of the tropical forests. They represent 75 % of the tropical coastal vegetation (in Mougin *et al.* 1999), which is of particular concern as unfortunately mangrove forests are one of the ecosystems that are being degraded and lost at very high rates in some regions of the world (2.5% yr^{-1}) so that less than 50% of their original area remains today (Duke *et al.* 2007). Mangroves are distributed in 123 countries and territories in the tropics and subtropics (Spalding *et al.* 2010); therefore, their adequate management is paramount while supporting local sustainable livelihoods.

Although several international conventions and national agreements have been signed to promote the conservation and sustainable use of forests, wetlands and biodiversity, such as the Convention on Biological Diversity (CBD) and the Ramsar Convention, biodiversity and habitat loss, as well as ecosystems degradation rates are high, reducing the capacity of ecosystems and forest to provide ecosystem services that benefit humans. Recently, it has been recognized through the CBD the importance of advocating integral ecosystem management approaches to mainstream biodiversity conservation in forestry, agriculture and tourism activities. Therefore, it is paramount to promote and evaluate the sustainability of forest management practices, as well as the sustainable management and use of natural capital, as different forestry approaches may result in different biodiversity conservation outcomes. The minimum requirement to achieve sustainability is the maintenance and enhancement of biodiversity and natural capital (Costanza and Daly 1992), and the achievement of better social conditions for all humans (UNDP 2011).

The environmental, and biological effects of different wood production methods in mangrove ecosystems are not widely studied. One of the main challenges to accurately assess their effects is to determine adequate sustainability criteria and indicators. While it has been proposed current natural capital stocks as the minimum limit to achieve sustainability (Costanza and Daly 1992), in the case of forests, this could be somehow ambiguous as current natural capital stocks of some forest may not be at their maximum capacity due to overharvesting and ecosystem degradation. Therefore, the link between multiple ecosystem services and natural capital has to be addressed to achieve the Sustainable Management of Forest Ecosystems. The ecosystem management for the production of only one ecosystem service, such as wood, often results in substantial declines in the provision of other ecosystem services (Bennet *et al.* 2009).

In the case wood production, several practices are considered sustainable based on the maintenance and regeneration of timber stocks in long-terms periods of time regardless of the silviculture method applied and the effect on biodiversity. However, with only one indicator it cannot be assumed that the maintenance and conservation of the full pool of ecosystems services and goods obtained from forests. Not considering the size and biological importance of the area used for wood production may affect landscape connectivity, reduce habitat for endangered species and biodiversity of a given area, as well as several other ecosystems services. While a wood production scheme could be sustainable from timber perspective, it may not be viable to maintain the ecological function of a system. Therefore, the importance of conducting sustainability forestry studies considering a biodiversity and conservation approach.

For this study, I evaluated forest structure attributes of the most intensively and recently harvested mangrove stands, and the most conserved mangrove stands to assess the sustainability of the small-scale harvesting activities in the Management Unit for Wildlife Conservation (UMA) of the Ejido San Blas in West Mexico. Five primary forest structure attributes were considered: stem density, canopy cover, diameter size classes, species composition, natural regeneration, basal area and wood volume. Forest structure attributes can be used as proxies to determine and evaluate effects, resilience and responses in mangrove forests to recent and historical continuous natural and anthropogenic disturbance regimes, such as hurricanes and logging.

5.2. Mangrove Forest Structure and the Resilience of the Forest to Harvesting

Mangrove forest structure is influenced by several internal-external abiotic and biotic factors (Tomlinson 1986; Lovelock *et al.* 2005; Berger *et al.* 2008), as well as hurricanes and anthropogenic disturbances. The magnitude and periodicity of such factors and the frequency of storms are important (Lugo and Sneadeker 1974; Cintron and Schaeffer-Novelli 1984; Tomlinson 1986). While large canopy gaps post-hurricane can be frequent due to patches created as the result of high salinity concentrations, large tree wind-throw and natural mortality of trees, which could result in slow mangrove recovery (McKee *et al.* 2007; Milbrandt *et al.* 2006), some times years and decades, small light gaps could enhance natural regeneration, seedlings growth rates, densities and establishment (in Krauss *et al.* 2008). Mangrove forests that develop in riverine coastal zones, such as the San Blas Mangrove System, tend to have the largest canopies and DBHs and are more structurally developed (Pool *et al.* 1977). However, the development of mangroves subject to frequent storms and hurricanes, such as those in this study could be hindered, resulting in either even-aged or uneven-aged forests, small to medium DBHs and high tree densities. From diameter size analysis one can assume that San Blas Mangrove Forests are uneven-aged. Differences were recorded in the frequency of individuals within DBH size class with higher densities in small DBH in both harvested and non-harvested stands.

San Blas mangrove communities are composed of three main true mangrove species: *Rhizophora mangle L.* (Red Mangrove), *Avicennia germinans,* (L.) L. (Black mangrove) and *Laguncularia racemosa*, with the secondary presence of *Conocarpus erectus.* The dominance of L*aguncularia* in the San Blas Mangrove System may be attributed to its resilience to hurricanes and wood harvesting. It has been documented that in canopy gaps, density and growth rates of *Laguncularia,* saplings tend to be greater than those of *Avicennia* and the former has lower mortality rates (Sherman *et al.* 2000). In this study, although average *Laguncularia* tree density was 42% higher in harvested stands this was not a significant. Lower tree density likely did not occur as a result of harvesting, as harvesting could stimulate vegetative reproduction through coppicing of *Laguncularia*. But, the lack of difference could be also the result of damage from Hurricane Kenna in 2002 in non-harvested stands. However, non-harvested stands had a higher average tree height than harvested stands, although average height in the harvested stands (10.7 ±1.9 m), represents medium to tall trees, reflecting sustainable wood harvesting. The San Blas harvested mangrove forest stands have similar, height, DBH, and basal areas but higher densities that those reported for unmanaged non-harvested riverine mangrove forests dominated by adult *Avicennia* trees in French Guyana (Fromard *et al*. 1998), and similar basal areas and tree height as in non-harvested riverine mangrove forests in the Pacific of Costa Rica (Pool *et al*. 1977).

At San Blas average basal area (24.8 m^2 ha^{-1}) is similar to that reported by Kovacs *et al.* 2001a for non-harvested forest before the impact of Hurricane Rosa (23.15 m^2 ha^{-1}), and more than double that reported after the hurricane (11.23 m^2 ha^{-1}). Tree height average in regulated managed harvested stands in San Blas is higher than the values reported for an unregulated harvested peri-urban mangrove forests in Kenya, which are subject to forest structure stress due to enlarged canopy gaps (Mohamed *et al.* 2009). Overall, mangrove forest structure measured in this study suggests that the San Blas mangrove forests may have a high resilience to both hurricanes and wood harvesting. As documented here, the San Blas mangrove forests had higher tree densities than in many un-harvested and harvested forests elsewhere, even after being damaged by a large-scale hurricane. As well as basal areas similar to un-harvested stands, and higher basal areas in comparison to some mangroves also affected by the impact of hurricanes.

5.3. Natural Regeneration

There are no generally accepted criteria and indicators to define what could be an adequate natural regeneration in mangrove ecosystems. Natural regeneration in Mexican mangrove forests ranges from 143 to 600,983 ind. ha^{-1} (Table 12) and numbers of seedlings and saplings reported for different mangrove systems in the world averaged 37,538 ind. ha^{-1} (Table 12 and 13). Therefore, natural regeneration is highly variable within and among mangrove systems, as are the natural biotic and abiotic, and anthropogenic factors that influence natural regeneration and mangrove forest structure. Furthermore, approaches to assessing natural regeneration are also highly variable. An additional complication is that the definition of seedlings, saplings, and trees varies among studies. In the majority of studies individuals with DBH >2.5 cm are considered trees (Pool *et al.* 1977; Valdez-Hernández 2002a; Bandeira *et al.* 2009; Mohamed *et al.* 2009; Rocha-González *et al.* 2012; Téllez-García and Valdez-Hernández 2012), while some define trees with DBH >5 cm (e.g. Sherman *et al.* 2001) or more >10 cm.

The majority of studies classify natural regeneration among classes, but the attributes of classes vary. Some considered natural recruitment as seedlings and saplings <130 cm (Rocha-González *et al.* 2012). Others have used different parameters. For instance, some considered Class I as seedlings <40 cm and Class II as seedlings 40 to 150 cm (Bandeira *et al.* 2009;

Mohamed *et al.* 2009). However, the major difference is designation of Class III, or saplings (See Table 1 in Chapter 2 for more details).

Natural regeneration in San Blas was assessed considering three height classes following the classification and similar methodology applied by Valdez-Hernández (2002a). Two categories of seedlings Classes I (< 30) and II (> 30), and one category for saplings Class III (> 130). In this study, saplings were >130 cm while others considered heights up to 300 cm as saplings. Overall, there was no significant difference in mean natural regeneration of seedlings (Class I and Class II) within each tidal zone between harvested and non-harvested. In addition, average tree density did not differ between harvested and non-harvested stands therefore; similar natural regeneration can be expected. The number of *Laguncularia* seedlings in Class I was significantly higher in harvested stands than non-harvested, while the seedling numbers in Class II were not significantly different. While saplings (Class III) were only recorded in non-harvested stands, saplings were observed in high densities in harvested stands outside the sample plots, suggesting that regeneration may be underestimated in this study. However, high densities of young trees >2.5 and < 10 cm were recorded suggesting adequate sapling establishment.

At San Blas *Laguncularia* seedlings average and range (800 to 30,000 ha^{-1}) post hurricane Kenna (2002) were higher than reported post hurricane Kenna by Rocha-González *et al.* (2012), but lower than reported by Valdez-Hernández (2002a) pre hurricane Kenna for a neighbor mangrove ecosystem in Villa Juarez. Kovacs *et al.* (2001a) reported low natural regeneration of *Laguncularia* (the dominant species) post hurricane in the Teacapán-Agua Brava Lagoon, West Mexico, which was severely damaged by hurricane Rosa in 1994, suggesting that environmental changes post hurricane such as light, substrate and topography hindered the recovery. In contrast, at San Blas a major hurricane did not hinder the regeneration capacity of *Laguncularia*.

Table 12. Ranges and averages of tree and juveniles densities of mangrove forests in West Mexico.

Country	Site	Juveniles No ind. ha^{-1} Ranges	Juveniles No ind. ha^{-1} Avg.	Tree density No ind. ha^{-1} Ranges	Tree density No ind. ha^{-1} Avg.	Mangrove Forest Condition	Forestry Plan	Mangrove Species	Ref.
Mexico	San Blas Mangrove System, Southern Nayarit	800 to 30,000	10,262	1,567 to 2,333	1,978	Conserved/non-harvested. Hurricane damaged (Kenna 2002)	yes	*L. racemosa* *R. mangle* *A. germinans*	This study
		800 to 30,000	12,160	3,233 to 3,533	3,389	Sustainable harvested for cultural traditions, domestic, and commercial proposes. Hurricane damaged (Kenna 2002)	yes		
Mexico	Villa Juarez Mangrove System, Southern Nayarit	625 to 600,938	147,085	699 to 8,132	3,646	Harvested	yes	*L. racemosa*	Valdez-Hernández 2002a
		111,687 to 395,587	223,408	1,267 to 10,632	4,473	Non-harvested			
		12,938 to 70,312	33,300	466 to 5,965	1,886	Harvested		*A. germinans*	
		625 to 138,300	60,132	565 to 7,267	2,286	Non-harvested			
		625 to 600,938	104,416	466 to 8,132	2,766	Harvested		*L. racemosa* *A. germinans*	
		625 to 395,587	121,361	565 to 10,632	3,379	Non-harvested			
		Seedlings 3,700 to 9,200	-	-	-	Harvested/non-harvested, Hurricane damaged (Kenna 2002)	-	*L. racemosa* *R. mangle* *A. germinans*	Rocha-González *et al.* 2012
Mexico	Teacapán-Agua Brava Lagoon System Northern Nayarit	Seedlings 5,500 to 103,875	-	-	-	Some communities in the area may have permits to harvest wood, Hurricane damaged (Rosa 1994)	-		Kovacs *et al*. 2001a
Mexico	Cuyutlan Lagoon, Colima	143 to 624	364	1,867 to 2,831	2,220	Non-harvested, unmanaged. Impacted by hydrological modifications	no		Téllez-García and Valdez-Hernández 2012

Table 13. Ranges and averages of tree and juveniles densities of mangrove forests in the world. Asterisk (*) indicates average values.

Country	Site	Juveniles No ind. ha^{-1}	Tree density No ind. ha^{-1}	Mangrove Forest Condition	Forestry Plan	Mangrove Species	Ref.
Mozambique	Saco and Sangala, Inhaca Island Southern Mozambique	*14,766	*1,966 to 3,680	Natural unmanaged and over-harvested forests	no	*A. marina, R. mucronata S. alba C. tagal*	Bandeira *et al.* 2009
	Mecufi, Pemba, Ibo Island, Luchete,Ulo (Northern Mozambique)	*14,706	2,080 to 2,753		no		
Tanzania	Mngoji	*2,212 to 4,799	1480 to 2286		no		
Kenya	Uyombo	Seedlings (0-40 cm) *183,344 Saplings (40-150 cm) *49,186	*1,585	Unmanaged but exploited not pristine mangroves for wood extraction	no	*A. marina B. gymnorrhiza C. tagal L. racemosa R. mucronata*	Kairo *et al.* 2002a
	Kirepwe	Seedlings (0-40 cm) 77,400 Saplings *21,478 (40-150 cm)	*1,197		no	*A. marina B. gymnorrhiza C. tagal R. mucronata*	
	Tudor Creek, Kombeni River (Mombasa Island)	*33,953	*1,264	Natural unmanaged but exploited mangroves for wood extraction	no	*R. mucronata C. tagal A. marina B. gymnorrhiza S. alba*	Mohamed *et al.* 2009
	Tudor Creek, Tsalu River (Mombasa Island)	*21,605	*1,301		no	*R. mucronata C. tagal A. marina B. gymnorrhiza S. alba X. granatum*	
Federated States of Micronesia	Kosrae Island	*17,600	-	Natural mangrove forests	no	*B. gymnorrhiza S. alba R. apiculata*	Pinzon *et al.* 2003
		*9,000	-	Natural unmanaged mangrove forests used to obtain firewood			
Indonesia	Cimanuk river delta, Java Island	Seedlings: 5,268 to 73,500 Saplings: 5,268 to 5,660	-	Natural expanding mangrove forests	-	*A. marina A. officinalis A. corniculatum B. parviflora R. apiculata R. mucronata*	Sukardjo *et al.* 2014

As this study was conducted after the impact of a large hurricane in the area that severely impacted San Blas Mangrove Forests, it is possible that the natural regeneration potential of San Blas Mangrove Forests is not at its maximum, but it could be considered that it is adequate for a post major hurricane event. Nevertheless, the results of this study support the main hypothesis of this study that harvesting activities do not reduce the capacity of the forest to regenerate; however, the hypothesis that natural regeneration is higher in harvested stands is not supported with the results of this study. In this study it is considered that natural regeneration in harvested stands is underestimated, as harvesting may contribute to enhance natural regeneration as the removal of timber on a selective basis creates small gaps within the forest stand. Pinzon *et al.* (2003) noted that gaps resulted from harvesting enhanced natural regeneration in unmanaged Micronesian mangroves frequently harvested mainly for firewood for domestic proposes. The results also reveal that the impact of a large-scale hurricane in the zone does not reduced the capacity of the forest to regenerate and that its resilience to human activities and natural events is possibly high. Duah-Gyamfi *et al.* 2014 found similar results in terrestrial forests, that logging does not weaken forest tree regeneration and can contribute to enhance regeneration at the initial stages of the harvesting until canopy closure is reached, which seems to be the case of San Blas.

While in some systems high regeneration may be recorded, low establishment rates may be occurring resulting in low tree densities and vice versa. Regeneration in the harvested forests of San Blas is similar to that reported by Pinzon *et al.* 2003 for natural mangrove forests on Kosrae Island in Micronesia. The natural regeneration at the San Blas Mangrove System is also similar to that reported for natural expanding mangrove forests in Indonesia (Sukardjo *et al.* 2014) and for unregulated harvesting in Mozambique, but higher than what has been reported for unregulated and over-harvested mangrove forest in Tanzania (Bandeira *et al.* 2009) and for non-harvested forests impacted by hydrological modifications (Téllez-García and Valdez-Hernández 2012). Mohamed *et al.* (2009) reported higher seedling densities in the Mombasa Island in Kenya than at San Blas, but concluded that seedling survival would be limited due to the presence of large canopy gaps that resulted from harvesting, as it was suspected that the large gap size could cause high soil salinities, which would limit the survival of seedlings. Kairo *et al.* 2002a found patterns similar to that found at San Blas, as the frequency of small trees was higher than large ones at Uyombo and Kirepwe, with overall regeneration among three regeneration classes 86:51:1 and 62:17:1, respectively. At San Blas ratios among three regeneration classes were 2.5:2:1.

5.4. *Laguncularia* Tree Diameter Distribution Analysis by 2 cm Size Classes

Disturbances resulted from both human activities and natural disasters in mangrove forests can affect forest structure and natural regeneration due to cumulative impacts (e.g. Kairo *et al.* 2002a). Diameter distribution can be used as a stand forest structure index to evaluate the sustainability of forestry activities, such as wood production, but also contribute to evaluating the resilience of mangrove forests to hurricanes. Therefore, DBH size class analysis conducted in this study may also contribute to the evaluation of the resilience capacity of San Blas Mangrove Forests after the impact of Hurricane Kenna. Natural mortality and disturbance from pest and hurricanes are the principal regulators of DBH size classes in natural forests (e.g. Chazdon *et al.* 2007). Diameter distribution and tree density could be hindered, maintained or enhanced through harvesting, limiting or allowing the re-population of a harvested stand through each cut.

Depending on the management approach applied to produce wood, harvested stands could be either even-aged or uneven-aged. Diameter distribution analyses by 2 cm size classes revealed that harvested and non-harvested forest stands at the San Blas Management Unit are uneven-aged. In mangrove ecosystems it could be difficult to determine tree age due to the limited development of tree rings and morphological plasticity due to micro-environmental conditions (Tomlinson 1986; Farnsworth and Ellison 1996; Feller *et al.* 2010); however, stands are considered uneven-aged following local villagers information regarding stand age and harvesting rotation schemes, as well as due to the variety of tree diameters registered. Production of an uneven-aged forest requires adequate regeneration on a yearly basis (Wittwer *et al.* 1990), which seems to be the case at San Blas. Currently, uneven-aged silviculture is considered at global scales an efficient alternative to even-aged approaches that may hinder landscape aesthetics, resilience to climate change and other natural phenomena, wildlife management, biological diversity conservation, and maintenance of canopy cover and multiple forest ecosystem services (IUFRO 2015). Long-term studies have shown that in European forests this system ensures a regular supply of timber and improves stands quality over time (e.g. Pukkala *et al.* 2009, 2010). In Mexico, even-aged approaches are forbidden in mangrove ecosystems (SEMARNAT 2012).

When small-scale selective harvesting is applied, as in the case of San Blas, uneven-aged forests similar to unmanaged natural forests can be obtained. Selective harvesting is the silvicultural process that generates and maintains uneven-aged forest stands, through the simultaneous removal of individuals or groups of mature trees to allow the regeneration of a new class and the thinning of young age classes (UF 2015). However, an uneven-aged stand-alone does not reflect the sustainability of the harvesting, and therefore it is critical to evaluate the distribution and frequency of trees among DBH size classes. Besides, selective small-scale harvesting can have significant impacts on forest structure and the ecological function of forests depending on the intensity, method, and tools used to remove timber.

Uneven-aged forests are characterized by having more than three age-classes with high amounts of young individuals versus old-growth trees. Three groups of age classes that are important to consider in an uneven-aged forest are: 1) a regeneration class, which includes seedlings and saplings, 2) timber of commercial size which may vary locally, and 3) mature and old-growth individuals as parental trees. These three groups were registered within the forest stands evaluated in the San Blas Mangrove Management Unit. Research on the structural dynamics, stocking control and the notion of a sustainable equilibrium state of uneven-aged forest stands are extensive for terrestrial forests (e.g. O'Hara and Gersonde 2004; Pukkala *et al.* 2009, 2010), but less is known for mangroves. Adams and Ek 1974, highlighted that two enduring challenges for the management of uneven-aged forests are: 1) determine the optimal sustainable distribution of trees by diameter class and 2) the optimal cutting cycle for the conversion of an irregular stand to a target structure. To achieve sustainable forestry practices and to assess if natural regeneration and tree distribution by size classes are maintained and adequate to sustain further harvesting conducting long-term diameter size class analysis is paramount.

In several mangrove systems, it has been reported that during large-scale hurricane disturbances the most vulnerable trees are those with the largest diameters (e.g. Kovacs *et al.* 2001b, 2004), which tend to be the first to fall affecting in many cases the resilience/regeneration capacity of the forests. For instance, Kovacs *et al.* (2001a) conducted DBH size class analysis to evaluate hurricane impact on mangrove forest structure in the northern coastal area of Marismas Nacionales after the impact of Hurricane Rosa 1994. They found that 80% of the trees with DBH larger than 20 cm were dead, while only 6% of trees with small diameters (2.5-5 cm) were affected. They reported a reduction of 31% of tree densities (from 3,520 to 2,423 ind. ha^{-1}) in non-harvested forests. The decrease in density due to the loss of large trees is consistent with the results of this study for the San Blas Mangrove System. Although at San Blas, trees in non-harvested stands had larger DBHs, their frequency was not high in comparison to young trees and to other natural mangrove forests. Local villagers have attributed the low abundance of trees with larger DBH in both forests stands (harvested and non-harvested) to Hurricane Kenna 2002. During fieldwork, several large dead trees were observed on the forest floor, mainly in non-harvested areas. Therefore, in non-harvested areas Hurricane Kenna may have caused the wind-throw of large trees and this may help explain why average tree densities are lower than reported by Valdez-Hernández (2002a) for pre-hurricane conditions in a neighboring mangrove system.

Adequate natural regeneration from seeds occurs depending on the biology of the commercial species harvested, the management and harvesting approach, as well as local conditions. Multiple attributes of mangrove forest structure and tree diameter analysis have been used to evaluate the sustainability of harvesting in several regions of the world, including natural regeneration. For instance, Bandeira *et al.* 2009 studied unmanaged harvested mangroves by local communities in Mozambique and Tanzania using DBH size classes. Overall, Bandeira *et al.* 2009 reported different levels of exploitation within both Mozambique and Tanzania, with low regeneration in some sites, indicating a need for improving mangrove management in these areas.

At San Blas, the modal diameter in harvested stands was 5.1 cm, and the most frequent diameter size classes were between 2.5 and 10 cm, two to three times higher in harvested areas. Reflecting perhaps a better regeneration pace than non-harvested forest bolstered by harvesting through allowing the creation of temporal small light gaps that benefits seedlings establishment (e.g. Feller and McKee 1999) and through enhancing the capacity of *Laguncularia* to resprout by coppicing (*personal communication with local villagers*). Diameter distribution and tree density could be maintained or enhanced through selective harvesting allowing the re-population of a harvested stand and virtually constant yields from each cut. Kairo *et al.* (2002a) observed that in an unmanaged harvested forest in Kenya the species most valuable for harvesting significantly declined. Despite the annual harvest of *Laguncularia*, this study showed that high tree densities, adequate canopy cover and natural regeneration of *Laguncularia* are occurring. *Laguncularia* is the dominant species in the evaluated areas, but there are areas dominated by *Rhizophora,* mainly borders of water bodies, and by *Avicennia* in more saline soils. Therefore, at least in the case of San Blas Mangrove System the dominant species and the most important commercially is maintained at adequate levels post-hurricane.

In San Blas, only trees with DBH >8 cm are harvested. According to this study, high densities of residual trees (> 2,988 trees ha^{-1}) with diameters from 2.5 to 8 cm left after harvesting in San Blas will eventually reach large DBH sizes for the next thinning cycles. Valdez-Hernandez 2004 proposed rotation periods from 8 to 16 years for San Blas, but if wood volume increase is high, trees may be harvested in shorter periods of time between 8 and 10 years (*personal communication with local villagers*). In harvested stands trees were observed with DBHs up to 34 cm. Mature and old-growth trees will serve as parent trees, enabling natural regeneration by seedlings in addition to the vegetative reproduction through coppicing. Tree diameter distribution and frequency analysis indicated that *Laguncularia* has a high enough natural regeneration capacity to secure the quantity of established young trees that could be harvested in the future unless a natural disaster or a disturbance occurs.

Table 14 compares forest structure attributes recorded in this study to those reported for Villa Juarez (Valdez-Hernández 2002a; Rocha-Gónzalez *et al.* 2012), a mangrove system located only 20 km north from San Blas, where local communities also harvest mangroves. At both sites, according to diameter size classes, there is an uneven-aged distribution of trees. At San Blas 12 classes were shared and not significantly different between harvested and non-harvested, similar to results of Valdez-Hernández (2002a) who concluded that there were no significant differences in forest structure among conditions. Only small differences in the diameter size classes represented were observed between San Blas and Villa Juarez, which may be the result of several factors, such as differences in natural forest structure, micro-environmental conditions or the impact of Hurricane Kenna in 2002. Canopy cover at San Blas is also similar to Villa Juarez (Rocha-Gónzalez *et al.* 2012), which corresponds to moderately to very dense mangrove forests.

Table 14. Forest structure of mangrove forests dominated by *Laguncularia* in West Mexico.

Communal landholder "Ejido"	Forest condition	# 300 m^2 plots	No. DBH size class	DBH size class not represented	Tree density (trees/ha)	Basal area	Canopy cover (%)	Ref.
Ejido San Blas	Harvested	3	14	20, 22, 36, 38	3,389 ±150	17.1 ±3.0	66 ±7	This study
	Non-harvested	3	16	2, 30	1,978 ±386	28.7 ±2.1	57 ±8	
	Ecosystem	6	18	26	2683 ±816	22.9 ±6.8	61 ±9	
Ejido Villa Juarez	Harvested	5	15	18, 28, 30	4,473 ±3,941	18.7 ±8.7	-	Valdez-Hernández 2002a; Rocha-Gónzalez *et al.* 2012
	Non-harvested	5	14	26, 32, 34, 36	3,646 ±3,255	23.0 ±12.0	-	
	Ecosystem	10	18	-	4,059 ±3,435	20.9 ±10.0	56 ±21	

5.5. Mangrove Standing Wood Volume

Current standing timber volume is an important indicator to determine the sustainability of mangrove wood production. At San Blas, in a surface of 0.18 hectares, wood volume ranged from 96 to 152 m^3 ha^{-1} in harvested mangroves and from 187 to 224 m^3 ha^{-1} in non-harvested, which are considered high in comparison to the values reported for other mangrove systems elsewhere and to the average wood volume reported pre-hurricane (Valdez-Hernández 2004) (Tables 15 and 17). Valdez-Hernández (2004) reported for 307 ha in San Blas wood volume ranges from 62 to

168 m^3 ha^{-1} before the impact of Hurricane Kenna in 2002. Average wood volume reported in this study, suggests that the San Blas Mangrove System, may not just be resilient to long-term harvesting activities but also large-scale hurricane impacts.

Table 15. Standing commercial wood volume (m^3 ha^{-1}) of *Laguncularia* in the San Blas Mangrove System before and after the impact of Hurricane Kenna.

Mangrove forest condition	Hurricane Kenna 2002	Studied Surface (ha)	Commercial Wood Volume (m^3 ha^{-1})			Reference
			Average	Min.	Max.	
Harvested	After	0.18	114.7 ±31.0	96.2	151.5	This Study
Non-harvested			209.3 ±19.7	193.7	230.8	
Ecosystem			162.0 ±57.0	96.2	230.8	
Ecosystem	Before	306.9	86.7 ±31.5	62.4	168.4	Valdez-Hernández 2004

5.5.1. Historical Wood Volume Changes in the Management Unit of the Ejido San Blas

Wood volume estimations and natural regeneration at the plot level were scaled up using the results of historical mangrove cover changes to determine the long-term sustainability of harvesting activities in the Management Unit for Wildlife Conservation of the Ejido San Blas (Table 16). From 2005 to 2010 net mangrove cover increased by 2.7 ha^{-1} yr^{-1} while the authorized average logging intensity was 6 ha^{-1} yr^{-1}. However, only 4.2 ha were lost during this period, and surprisingly, there was a net gain of 13.6 ha, more than double the authorized harvesting quota. According to local villagers, logging intensity was greater from the 1930s to 1990s. Considering an hypothetical logging average of 1,400 m^3 yr^{-1} for the period 1981-2005 and the average wood volume estimated in this study, a loss of at least 12.2 ha^{-1} yr^{-1} or an equivalent of 292.8 ha should have been expected in a period of 24 years, but the loss estimated for this period was only 44.2 ha, and the gain was 324.3 ha, resulting in a net gain of 280.2 ha.

Overall, wood socks in the Management Unit for Wildlife Conservation of the Ejido San Blas have been increasing with the highest rates in Conservation Zones followed by Protection

Zones. Wood stocks in production zones #2 and 3 increased since the period 1970/1980, but decreased by 179 m^3 in Zone #1. From 2005 to 2010 wood stocks were stable in Protection and Production Zones but increased in Conservation Zones by 2,857 m^3. These mangrove cover increases in conserved and protected zones, and the maintenance of mangrove cover in harvested stands indicate that applying zoning approaches for the management of mangrove forests for wood production through Management Units for Wildlife Conservation are effective for mangrove forests conservation.

Table 16. Total standing wood stocks (m^3) at different temporal and spatial scales at the Management Unit for Wildlife Conservation of the Ejido San Blas.

Year	Total Stocks in the UMA	Protection Zone	Conservation Zone	Production Zone #1	Production Zone #2	Production Zone #3
1970/80	198,087	11,405	162,861	7,021	10,187	6,615
2005	255,098	13,808	215,495	6,842	11,784	7,169
2010	257,955	13,808	218,352	6,842	11,784	7,169

5.5.2. Standing Wood Volume of Mangrove Systems in the World

From the few available studies, I calculate that the global average standing wood volume in mangrove ecosystems is 157 m^3 ha^{-1} and ranges from 78 to 244 m^3 ha^{-1} (Table 17). In San Blas, maximum wood volume in harvested stands (152 m3 ha^{-1}) is similar to the global average (161 m3 ha^{-1}) estimated in this study, and maximum wood volume in non-harvested stands (224 m3 ha-1) is higher than the global average. Téllez-García and Valdez-Hernández 2012 reported low average values in an unmanaged stressed mangrove system in Colima, West Mexico. Average wood volume reported for the most intensively and recently harvested stands in San Blas coincide with the wood volume reported in other regions of the world, including the Matang Reserve in Malaysia (Haron, 1981 in Kairo *et al.* 2002a). Standing wood volumes in San Blas also coincide with values reported in Sumatra (Sukardjo 1987), Kenya and Indonesia (Kairo *et al*. 2002a,b).

Table 17. Standing wood volume ($m^3\ ha^{-1}$) of mangrove systems in the world.

Country	Site	Wood Volume ($m^3\ ha^{-1}$) Min	Max	Avg	Average Densities per ha	Mangrove Forest Management Approach	Forestry Plan	Mangrove Dominant Species	Ref.
Mexico	San Blas, Nayarit	96	152	115	3,389 (DBH >2.5cm)	Community-based managed for wood production	Yes	*L. racemosa*	This study
		187	224	209	1,978 (DBH >2.5cm)	Community-based managed for conservation	Yes	*L. racemosa*	
		62	168	87	- -	Regulated community-based harvesting	Yes	*L. racemosa*	Valdez-Hernández 2004
Mexico	Barra de Tecoanapa, Guerrero	-	-	96	1,175	Unregulated local harvesting of *Laguncularia* and *Rhizophora* for domestic use	No	*L. racemosa* *R. mangle* *A. germinans*	Tovilla-Hernández *et al.* 2001
Mexico	Cuyutlan Lagoon, Colima	23	69	45	2,414 (DBH >2.5cm)	Unmanaged and un-harvested forest	No	*L. racemosa* *R. mangle* *A. germinans*	Téllez-García &Valdez-Hernández 2012
Costa Rica	Térraba-Sierpe	-	-	281	769	-	-	-	In Kairo *et al.* 2002a
Malaysia	Matang	-	-	153	1,343	Clear cut government managed	Yes	*R. apiculata*	
Indonesia	Ranong	-	-	226	812	-	-	-	
Kenya	Kiunga Marine National Reserve, Lamu	7	710	146	1,736 (DBH >5cm)	Unmanaged, over-exploited mangroves	No	*A. marina, R. mucronata, Sonneratia, Ceriops tagal*	Kairo *et al.* 2002b
Sumatra	Tanjung Bungin, Banyuasin district	-	-	251	226 (DBH >20cm)	Government designated for wood production. Local use for house construction.	-	*Rhizophora sp.* *Bruguiera sp.*	In Sukardjo 1987

There are several uncertainties when comparing wood volume among studies and sites. As different management approaches are applied, different species used, commercial tree diameter harvested, as well as different harvesting cycles considered in the management system. For instance, many of the studies reported in Table 17 used different DBH ranges to determine tree densities and wood volume ranging from >2.5cm to >20 cm. Besides, environmental conditions could vary greatly locally among mangrove forests.

5.6. Historical Mangrove Cover Change Analysis

Over the period of 1970/1980-2005 mangrove forests expanded in the Management Unit for Wildlife Conservation of the Ejido San Blas. From 2005-2010 mangrove cover was maintained, with almost no-loss and small increases observed. *Laguncularia* usually develops in areas where there has been increased sedimentation and lower influence of the tides (Pennington and Sarukhán 2005), which is the case of San Blas. Historical mangrove cover change analysis suggests that there were not natural or anthropogenic barriers during the evaluated periods limiting the landward expansion of mangrove forests within the Management Unit of the Ejido San Blas. However, in the East part of the location of the Management Unit, a road was constructed, with seems to limit the adequate development of mangrove systems.

Laguncularia and *Avicennia* may only need between 2-3 years to colonize a mud bank being pioneer species (Fromard *et al.* 1998). The coastal plains of the State of Nayarit are characterized by being fertile and dynamic due to flooding periods during the rainy season, which may be contributing to the increase sediment deposits and high nutrient availability from rivers, favoring mangrove expansion landward. In the case of Nayarit, the coastal plains are extensive and can reach up to 20 km inland allowing the development of extensive mangrove systems. On the contrary, in the neighbor state of Jalisco in the south of Nayarit geomorphology limits the development of mangrove ecosystems due to limited coastal plains as result of the concurrence of mountains and active tectonic movements in the area. The community-based management conducted by the Ejido San Blas has contributed to the conservation and expansion of the mangrove forests in the past 40 years. Berlanga-Robles and Ruiz-Luna 2006 reported mangrove losses for the periods 1973-1986 and 1986-1992 with a rate loss of 0.2% yr^{-1} in surrounding mangroves of the San Blas UMA due to agriculture and aquaculture activities. In contrast, important losses of mangrove have been occurring recently in surrounding areas due to the construction of aquaculture farms, which may limit the further expansion of mangrove forests in the area.

5.7. Community-based Mangrove Forestry: Implications for Management

Mangroves are important coastal social-ecological systems, as in most coastal regions local communities rely on the multiple ecosystem services mangroves provide them for their survival and livelihoods. Local communities living within mangroves are the primary beneficiaries of fish and wood products, which usually do not reach large-scale markets, reducing the recognition of these values at national and international levels. The current economic valuation of mangrove wood production represents just a minor part of all the services that mangroves provide, but it is a critical service at local levels. Environmental services "are a powerful lens through which to understand human relationships with the environment and to design environmental policy" (Brauman *et al.* 2007).

As documented in this study community-based mangrove wood production through Management Units for Wildlife Conservation could be a feasible and sustainable cost-effective conservation and management scheme for mangrove ecosystems. Furthermore, Management Units for Wildlife Conservation could be a win-win solution for the conservation of mangroves natural patrimony, its ecosystem services and biological diversity in the long term, within and beyond protected areas while providing sustainable livelihoods and achieving social, cultural, economic, and environmental benefits. While it is critical to prevent further losses and degradation of mangrove forests and ecosystems, for which laws and their appropriate application are needed, it is also important to promote inclusive and participatory management and use of mangrove forests through regulations and guidelines that could allow sustainable management and livelihoods (e.g. Kairo *et al.* 2001). It is in the best interest of local communities and indigenous peoples living within mangroves to contribute to the conservation and restoration of these ecosystems, as well as to implement forestry best management practices to improve and safeguard their livelihoods and traditional knowledge. Therefore, the success of community-based mangrove management highly depends on the recognition and safeguard of local communities and indigenous peoples traditional land tenure rights.

Community-based mangrove wood harvests occur worldwide; nevertheless, these activities are poorly documented, studied and regulated (Walters 2005a). This lack of knowledge has resulted in policies banning the traditional use of mangrove wood by local communities and indigenous people in some regions, as unregulated and unsustainable over-harvesting is considered an important driver of mangrove degradation (e.g. Valiela *et al.* 2001). More important, while the sustainability of community-based mangrove wood production is frequently questioned, few studies have evaluated the ecological effects of mangrove logging and its sustainability. The studies that have evaluated the effects of community-based harvesting in mangroves have produced mixed results.

Most of the available studies on mangrove wood harvesting by local communities report small-scale unregulated harvesting practices that are unsustainable. Besides, most of the studies that report either sustainable or unsustainable practices do not provide criteria and indicators to determine what is a sustainable or an unsustainable practice. Hence, evaluating the effects of different management and conservation approaches on mangrove ecosystem services could provide critical information for the development of criteria and indicators that can be used by decision-makers to draft and implement forestry and public policies that integrate community-based mangrove conservation and sustainable forest management. It may also contribute to the regulation and monitoring of both mangrove wood commercial production and mangrove harvesting for domestic use.

Mangrove harvesting, silviculture, and management approaches vary widely across regions, and in many cases, it could be sustainable as documented in this study, but more case studies and long-term monitoring programs are needed to promote mangrove best forestry practices. Documenting and communicating local communities and indigenous peoples' management and conservation approaches, as well as their socio-economic and cultural dependency on mangrove ecosystem services, may contribute to the promotion, maintenance, and enhancement of mangrove participatory stewardship and conservation following a sustainable forest management approach.

In the Gulf of Guinea, Cameroon, local communities harvest mangrove for fuel-wood and charcoal, which generates revenues of 400,000 Euros per year (Din *et al.* 2008). Timber harvesting is not conducted in a sustainable manner, as the logging intensity is at large-scale and conducted with out forestry plans. In a single day, it could be harvested the equivalent of 569 trees (Din et al. 2008), which is more than the double the volume authorized for a year in the San

Blas Mangrove System (Table 18). In Mozambique and Tanzania Bandeira *et al.* (2009) reported unregulated small-scale harvesting resulting in low tree densities and poor wood quality in harvested stands. In Timor-Leste, Alongi and de Carvalho (2008) described similarly unregulated harvesting where there was a reduction of 30 to 50% of tree density within a year. In the Island of Mombasa, Kenya, Mohamed *et al.* (2009) reported that unregulated wood harvesting resulted in large canopy gaps that structurally stressed mangroves. In the Philippines, Walters (2005a) observed large canopy gaps in harvested forests. Matang mangrove forest in Malaysia, where mangroves are clear-cut and planted following government-based management, is considered to have some of the best-managed mangroves in the world (Gong and Ong 1995; Alongi 2009; UNEP 2014). Interestingly, San Blas tree densities and natural regeneration reported in this study are higher than those of Matang mangrove forest.

Table 18. Community-based harvesting approaches reported in the world.

Country	Site	# Loggers	DBH	Thinning cycle	Production Area	Logging Intensity	Wood Volume	Forestry Plan	Ref.
			cm	years	ha	$m^3\ yr^{-1}$	$m^3\ ha^{-1}$		
Mexico	San Blas	50	8 - 12	8 to 10	228	700	96 - 224	Yes	This study
Mexico	Villa Juarez	-	7-12	5 to 7	270	540	-	Yes	Valdez-Hernández 2002a
Cameroon	Gulf of Guinea	350	-	-	900 - 1,100	171,550 (470 per day)	-	No	Din *et al.* 2008

Community-based mangrove forest management and harvesting activities conducted by the "Ejido San Blas" has allowed a permanent repopulation of the forests maintaining and enhancing wood volume for more than 70 years, as well as other ecosystem services, such as habitat for endangered species. Landscape connectivity and canopy cover in San Blas could be playing an important role in the resilience of the ecosystem to both harvesting and the impact of large-scale hurricanes as natural regeneration is maintained. In San Blas, after the removal of timber, leaves, twigs and wood debris resulted from the logged stems are chopped into small pieces and integrated into the forest floor, which may be contributing to enhancing organic carbon and nutrient inputs to the forest floor, as well as protecting seedling and saplings, which further improves natural regeneration. Woody debris can benefit the natural regeneration in mangrove swamps by trapping propagules and enhancing seedling growth potential by offering protection, as reported in other regions (Krauss *et al.* 2005).

Although *Laguncularia* standing commercial timber volume was significantly different between harvested and non-harvested stands, *Laguncularia* showed high wood productivity in harvested stands compared with other wood volume estimations reported elsewhere for natural mangrove forests. The harvesting and management approach applied by the community Ejido San Blas can be considered sustainable because after harvesting residual standing wood volumes are large compared to other unharvested and unmanaged natural mangrove forests, and uneven-aged forest stands are left. Besides, the authorized harvesting quota of 700 m^3 yr^{-1} is half of the maximum sustainable harvest of 1,400 m^3 yr^{-1} recommended by Valdez-Hernández (2004) for a rotation period from 8 to 16 years.

The Ejido San Blas applies 8 to 10-year thinning cycles; therefore, the wood volumes harvested are similar to those recommended by Valdez-Hernández (2004), but over shorter periods of time. Current harvesting volume is equivalent to removing 3 m^3 ha^{-1}, which eventually will be recover in short to intermediate time, as indicated and supported by natural regeneration and residual standing wood volumes reported in this study. The sustainability of wood production by the "Ejido San Blas" relies in part on the natural capacity of *Laguncularia* to resprout by coppicing and to the maintenance and enhancement of natural regeneration.

Furthermore, experienced harvesters use machetes and axes for tree removal, and commonly used natural paths within the forest to transport harvested trunks resulting in minimal damage to the forest floor from logging activities.

Community-based mangrove wood production could be an economically productive sustainable livelihood for local communities in many subtropical and tropical regions of the world, as well as in Mexico. Community-based forest management and traditional ecological knowledge have long been recognized as an essential element for the conservation and restoration of the remaining tropical forests, the sustainable use of natural resources and the provision of sustainable livelihoods (e.g. Hartshorn 1995; Merino *et al.* 1997; Huntington 2000; Usher 2000; Toledo *et al.* 2003; Charnley and Poe 2007; Bray *et al.* 2008; Porter-Bolland *et al.* 2012), but for mangrove ecosystems is not well established. However, community characteristics (i.e. size, composition, norms, resource dependence), along with institutional arrangements and processes of decision-making and enforcement may result in different natural resource management outcomes (Agrawal and Gibson 1999). Therefore, the importance of empowering local communities to achieve the sustainable management of natural resources and forest capital considering social benefits (Costanza 2003; Kuuluvainen *et al.* 2012).

The disturbances resulted from unregulated and unsustainable community-based wood harvesting, as well from agriculture can be reduced by designating small areas for wood production and by promoting the implementation of Community-based Mangrove Forest Management Plans. The latter could be based on traditional ecological knowledge, as well as on forestry and scientific assessments. The Sustainable Forest Management and Conservation of the remaining mangrove forests relies in part on the capacity of governments to recognize the importance of involving local communities in the management of mangrove ecosystems and in the decision-making process. Local community involvement could strengthen their capabilities while creating a co-responsible adaptive management approach. This study particularly highlights how biodiversity-rich and well-conserved mangrove forests can be managed by local communities and indigenous peoples, while generating income and supporting local markets. This has implications for the promotion and boost of best forestry practices, not only in Mexico but also in the world at large, as community-based forest management is an emerging field.

Therefore, the Ejido San Blas can be considered a successful community-based mangrove forestry case study that can be used as a model to promote forestry, conservation and management best practices and sustainable economic development at local scales elsewhere. Communities like the Ejido San Blas should be rewarded and considered for the achievement of several international agreements, such as the Aichi Biodiversity Targets of the Convention on Biodiversity Conservation and the Sustainable Development Goals. As well as to achieve the four global objectives on forests set by the United Nations Forum on Forests (UNFF 2016): 1) Reverse Forest Loss, 2) Enhance Forest-Based Benefits by improving the livelihoods of forest-dependent people, 3) Increase the Sustainability of Managed Forests, and 4) Mobilize Financial Resources for the implementation of sustainable forest management.

For the purpose of this study sustainable community-based mangrove forest management and use refers to the indicators shown in Table 19, considering forestry guidelines developed for the sustainable use of mangroves in the study area and including the results of this study (e.g. Valdez-Hernández 2002a; SEMARNAT 2012). Key factors of success of the wood production and conservation approaches applied by the Ejido San Blas can be summarized as follows: I) Landscape connectivity and the maintenance of hydrologic patterns, II) Community-based traditional ecological knowledge, III) Collaboration with scientists and forestry technicians, IV) Co-responsible management with the Government, V) Community-based monitoring and forestry studies, VI) Community organization and governance, VII) Community-based land tenure, VIII) Capacity-building. It would be important to further study the socio-ecological factors influencing the sustainability of wood production and the conservation of mangrove ecosystems in the Management Unit of the Ejido San Blas.

Table 19. Indicators to evaluate the sustainability of community-based mangrove wood production in Mexico.

INDICATORS	
1)	Uneven-aged silviculture.
2)	Selective tree removal limited to DBH size classes >8 cm. Selective tree removal in small surface production stands from 0.1 to 10 ha depending on the size of the mangrove area under management.
3)	Small-scale wood production per year, according to the capacity of the forest under management.
4)	Forest structure attributes such as wood volumes, tree density, basal area, natural regeneration and canopy cover (>60%) is maintained and enhanced.
6)	Internal creeks used to transport mangrove timber.
7)	Harvesting activities are conducted in areas easily accessed through water bodies, generally up to 100 m inland, leaving the first 20 to 30 m from the creeks untouched for water bodies protection.
8)	Around 80% of the surface area under community-based management is left conservation purposes.
9)	Use of low impact tools such as machetes and axes for logging. Transportation implemented with environmentally friendly approaches.

Drainage and degradation of organic soils (Armentano 1980), such as those of mangroves, can result in the release of large amounts of carbon stored over thousands of years to the atmosphere (De la Cruz 1986). One of the main environmental concerns regarding the use of mangrove forests for wood production is the release of high amounts of greenhouse gases resulted from soil degradation and loss. In this context, it is critical to promote community-based projects that consider regulated small-scale reduced-impact selective harvesting to allow long-term productive logging while maintaining and enhancing the carbon stored above and belowground (e.g. Miller *et al.* 2011), to prevent the decrease of biomass (e.g. Dyer *et al.* 2010).

Mangrove harvesting in Mexico requires the authorization from the government and currently, only a few local communities mainly in the State of Nayarit hold authorized UMAS for mangrove wood production. However, several communities continue to use mangroves for domestic purposes without Forestry Management Plans and authorized permits—an illegal activity under current legislation. In some cases strict conservation policies that prohibit the use of mangroves natural resources could constrain the preservation and wise use of the ecosystems, compromising local livelihoods. Still, it is imperative to conduct environmental and biological assessments before promoting and authorizing wood production to prevent biodiversity loss and habitat degradation, as each mangrove system is unique and provides ecosystem services at different scales (e.g. Bacon and Alleng 1992). Riverine mangroves as in San Blas, are dependant upon the sustainable management of forest and hydrological resources, as rivers supply sediments and fresh water to the system. Therefore, a watershed management is critical for mangrove functionality and development. Cautions should be taken when authorizing the construction of dams and roads, as well as river diversion for agriculture, as hydrological modifications can cause the degradation of mangroves and reduce their resilience to hurricanes, climate change, and sea-level rise.

Chapter 6. General Conclusions and Future Considerations

My thesis suggests that the Ejido San Blas mangrove forest management and wood production approach have been conducted sustainably resulting in reduced-impacts to the system. The sustainability of the wood harvesting approach conducted by the Ejido San Blas was assessed through estimations of stem density, natural regeneration, canopy cover, and commercial standing wood, as well as with tree diameter distribution and historical mangrove cover analysis. The mangrove silviculture approach applied by the Ejido San Blas is a small-scale reduced-impact selective harvesting that allows the conservation of 80% of the mangrove forests.

My primary hypothesis that community-based small-scale selective harvesting applied in the Management Unit of the Ejido San Blas does not reduce the capacity of mangrove forests to regenerate, maintain sustainable yields and landscape connectivity is validated. However, the data collected did not support my hypothesis that tree density and natural regeneration are higher in harvested stands, as both harvested and non-harvested stands had similar average values. Interestingly, average tree densities are high in harvested forests (3,389 ind. ha^{-1}) in comparison to other mangroves managed for wood production and to average global tree density values (2,400 ind. ha^{-1}) estimated in this study. Although *Laguncularia* average tree densities in both conditions were similar, stem density is likely to be underestimated in harvested stands.

The average number of seedlings was similar in harvested (10,178 ind. ha^{-1}) and non-harvested forest stands (13,422 ind. ha^{-1}). However, in harvested stands, *Laguncularia* natural regeneration is probably also underestimated. A larger sample area may have resulted in the measurement of saplings and better natural regeneration data collection, as outside the evaluated plots a high number of seedlings and saplings were observed. Nevertheless, results suggest that harvesting is enhancing natural regeneration and stem resprouting. Parental trees could be potentially available in both conditions until an external factor intervenes, such a hurricane. Although wood volume between conditions was significantly different, maximum wood volume in harvested stands (152 m3 ha^{-1}) is similar to the global average (161 m3 ha^{-1}) estimated in this study, and wood volume in non-harvested stands (224 m3 ha^{-1}) is higher than the global average. Supporting the hypothesis that the Ejido San Blas is maintaining sustainable yields of wood.

Tree density and canopy cover results supported my hypothesis that community-based small-scale selective mangrove harvesting does not reduce landscape connectivity. Although canopy cover ranged from 53 to 68% corresponding to moderately to very dense mangrove forests, the average canopy cover estimated was 60%, which is the minimum recommended in this study to maintain forest structure, but further studies may be needed to corroborate the value. Canopy cover was higher in harvested stands suggesting that community-based mangrove wood harvesting may contribute to enhancing the density and establishment of seedlings.

Historical mangrove cover analysis supported my hypothesis that mangrove cover in the Management Unit from 1970/1980 to 2010 was maintained. Indeed, the analysis revealed that mangrove forests in San Blas have been expanding. Mangrove expansion in the Management Unit can be the result of several factors including the protection of surrounding natural areas and limited development of infrastructure. Thus, allowing the migration, survival, and resilience of mangrove ecosystems. Overall, the results of my study suggest that mangrove forests located within the Management Unit may be resilient to wood harvesting and large-scale hurricanes.

My study did not address landscape heterogeneity due to geomorphologic, light, hydrologic, and edaphic variations, which may affect natural regeneration and forest structure. A focus on these through further research could be valuable. In this study, only one ecosystem service was evaluated "wood", further analysis to assess the effects of small-scale selective mangrove wood production on multiple ecosystem services should be conducted, such as the impact on habitat for endangered species and carbon sequestration. It would be important to consider dead trees standing and in the forest floor to estimate dead wood volume to assess mangrove forest structure and carbon stocks. Other aspects to contemplate are the quality of the standing wood and the number of resprouts and stems per tree, as well as litter fall. Although diameter analysis revealed a good representation of different tree development stages from young to mature trees with high frequencies in young stages, a long-term integral scientific monitoring program is recommended to determine the long-term sustainability of harvesting. More studies could contribute to establishing optimal regeneration ranges and tree DBH distributions by class to provide the guidelines for sustainable stem removal. It would also be important to evaluate and monitor tree growth and age among tree DBH size classes, as morphological plasticity related to environmental conditions at local scales may influence tree DBH distribution and frequency.

Community-based mangrove wood production through Management Units for Wildlife Conservation could be a feasible and sustainable cost-effective scheme for the conservation, restoration, and management of the remaining mangrove ecosystems within and beyond natural protected areas. Regulation of harvests through the designation of small areas for wood production and implementation of community-based mangrove forest management plans based on traditional knowledge and scientific assessments could reduce mangrove degradation caused by illegal small-scale harvesting and promote sustainable approaches. The Ejido San Blas should be rewarded and considered as a model for other local and indigenous communities. Community-based mangrove management could contribute to the achievement of multiple international agreements and conventions such as the Aichi Biodiversity Targets-CBD; the Sustainable Development Goals; The UN Forum on Forests' goals; the UNCCC and the Ramsar Convention.

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