The conservation status of mangroves and their contribution to artisanal fisheries in the Eastern Tropical Pacific

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Abstract

Mangroves are widely recognised as being one of the most valuable of coastal ecosystems. On a local scale they provide coastal protection, habitat for fish and shellfish, and control water quality. Globally, they are key to mitigate climate change given their considerable capacity to sequester carbon. In Latin America, especially in the Tropical Eastern Pacific region, mangrove forests are abundant, and linked to millions of coastal livelihoods. This thesis investigates the conservation status of mangroves on the Tropical Eastern Pacific coasts of Costa Rica, Colombia, Ecuador, and Panamá, by examining historical trends of mangrove loss, modern rates of deforestation (2000-2012), and the effectiveness of the protected area management in place. The contribution of mangrove-associated species to fisheries is evaluated using two artisanal fishing communities as case studies: Northern Chocó, Colombia and the Gulf of Montijo, Panamá. To do this, I investigate historical ecology, perform spatial analysis of proximate drivers of land use and land cover change adjacent to mangroves, and analyse small-scale fisheries landings. Results show that mangrove dependent species are important for small-scale fisheries in the Gulf of Montijo, Panamá. In Northern Chocó, territorial use rights in fisheries promote offshore fishing, thereby reducing fishing pressure on mangrove-associated species, but simultaneously may have displaced fishing effort from industrial trawlers into neighbouring areas. Regionally, mangrove area declined by almost 50% in the 20th century, but deforestation was virtually zero between 2000 and 2012, showing that protected areas are highly effective at conserving mangroves. Given that the success of mangrove conservation depends on government capacity to integrate multi-sectorial interests over mangroves, this thesis represents an important step to inform management strategies that involve a better understanding of human-mangrove interactions in Latin America.

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Acknowledgements

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For Chapter 3 Marviva Foundation and the University of Panamá provided artisanal fishery landings datasets for the Gulf of Montijo. Smithsonian Tropical Research Institute and Marviva Foundation also provided land use datasets. Supervised volunteers assisted me with my research by recording video interviews with fishers in some communities of the Gulf of Montijo that I was not able to visit during fieldwork.

For Chapter 4, I used artisanal fisheries landings data collected by Marviva Foundation in Chocó, Colombia.

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Memorandums of understanding for data sharing were signed with all collaborators when relevant.

I declare that this thesis is a presentation of original work and I am the sole author. This work has not previously been presented for an award at this, or any other, University. All sources are acknowledged as References.

Chapter 1. Introduction

1.1. Background information

The Eastern Tropical Pacific (ETP) biogeographical ecoregion, defined by Briggs (1974), spans the continental shelf and oceanic islands of Southern Baja California to northern Perú (Briggs & Bowen 2012). Within this region, the Pacific waters of Panamá, Costa Rica, Colombia, and Ecuador, have particular conservation significance given the presence of four World Heritage Sites: Cocos Island in Costa Rica, Malpelo Island in Colombia, Galapágos Islands in Ecuador, and Coiba Island in Panamá (Edgar et al. 2011). This area, which I refer to here as the ETP, encompasses 5,100 km from the continental coasts of southern Central America on the Nicaragua-Costa Rica border, to north-western South America on the Ecuador-Perú border (Figure 1.1). The ETP is very productive and biodiverse (Fiedler et al. 1991; Miloslavich et al. 2011) and sustains a high number of coastal communities plus some large cities such as Panamá City in Panamá, Buenaventura in Colombia, Guayaquil in Ecuador, and Puntarenas in Costa Rica, among others.

Figure 1.1. The Eastern Tropical Pacific encompasses the Pacific coasts of Costa Rica, Panamá. Colombia and Ecuador, highlighted in red.

1.1.1. Oceanography

The ETP lies at the confluence of the warm surface waters of South West Mexico, the equatorial current to the south, and the equatorial current system to the west of the Galapágos Islands (Fiedler & Lavín 2006). The Humboldt current in the south, and the California current in the north, divide the ETP Ocean into two regions that are dynamically and hydrographically distinct, namely: the Eastern Pacific Warm Pool, and the Equatorial Cold Tongue (Longhurst 2007). Water circulation in the ETP is an intricate process driven by oceanic and coastal forces, which are yet to be completely deciphered (Fiedler & Lavín 2006). This complex process is characterized by a number of currents (e.g. near-surface, upper-layer geostrophic, and subsurface), permanent eddies, and off-equatorial upwelling (Figure 1.2), which constantly interconnect to generate diverse seasonal change (Kessler 2006) and have an important role in the distribution of nutrients (Stramma et al. 2013). The varied and distinct oceanographic and climatic regimes have great influence on the abundance, distributions and life histories of pelagic and benthic communities (Chavez et al. 2003; Ballance et al. 2006; Fernández-Álamo & Färber-Lorda 2006). Circulation in the ETP is therefore very important to the region's complex ecology and climate.

The most important climatic influence in the ETP is the El Niño Southern Oscillation (ENSO), which is generated by the interaction between water temperature and air pressure of the Eastern and Western Tropical Pacific (Wang & Fiedler 2006). Notable climate patterns occur roughly every 4 years, where strong wind anomalies and a significant warming off the coast of South America create a condition known as El Niño, which in turn is followed by a cold phase called La Niña (Wang & Fiedler 2006). The ENSO phenomenon generates important variations in multiple processes including coastal upwelling, land and sea temperature, wind strength, circulation patterns, stratification, insolation, and hydrography (McPhaden & Yu 1999; Cai et al. 2014; Carre et al. 2014). These anomalies have important effects on productivity of phytoplankton and zooplankton, which in turn influence overall ecology of higher trophic levels (Wang & Fiedler 2006).

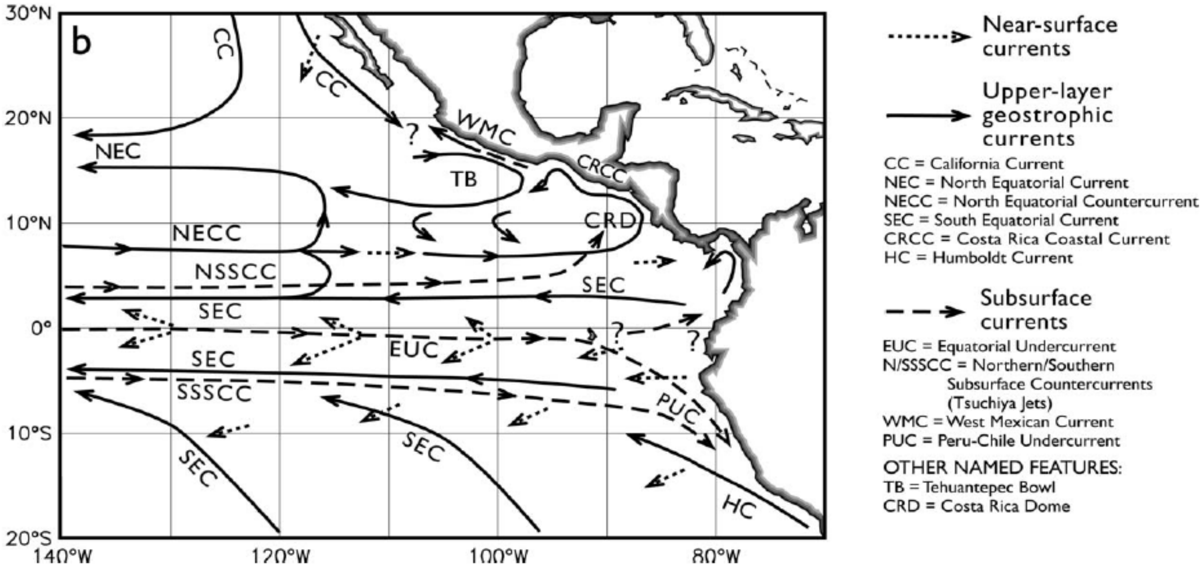

Figure 1.2. The complex circulation of the Tropical Eastern Pacific. The right hand legend provides the full names of the currents, while regions where interconnections of currents are unknown are depicted with question marks. Taken from Kessler (2006).

The numerous effects of ENSO on marine life have been well documented. For example, Toth et al. (2012) found that the increased variability of ENSO drove the collapse of coral reefs in the ETP for 2,500 years, beginning about 4,000 years ago. Barber et al. (1986) provided strong evidence for dramatic large-scale biological effects directly related to the 1982-83 El Niño. These include the redistribution of hake (*Meluccius gayi*) in Peruvian waters; changes in abundance of the shrimp *Xiphopenaeus riuetti, Penaeus occidentalis* and *Trachypenaeus byesi* from Colombia to Perú; rapid population increase of the scallop *Argopecten purpuratus*; absence of jack mackerel (*Trachurus symmetricus*) from Perú; decrease of sardine (*Sardinops sagax*) in Ecuador contrasted with its high abundance in Chile; and an increase in abundance of oceanic predators such as bonito (*Sarda chiliensis*), dorado (*Coryphaena hippurus*) and yellowfin tuna (*Thunnus albacares*).

One of the most significant examples of the biological effects of El Niño was its impact on the Peruvian anchovy (*Engraulis ringens*) fishery which collapsed in 1972 after warm waters from the El Niño greatly reduced the anchovy's planktonic food (Barber & Chávez 1986). The effect of this was exacerbated by years of previous overfishing (Clark 1977). Ten years later, the 1982-83 El Niño caused a 20-fold decrease in phytoplankton production, which caused severe nutritional stress for the anchovy and reduced its reproductive success (Barber & Chávez 1986). This illustrates how the synergy of environmental and biological factors led to the collapse of one of the world's largest fisheries, highlighting the need to account for environmental variability in fisheries management, particularly on heavily exploited stocks.

1.1.2. Habitat and biogeography

The first scientists to observe and study the ETP, including Charles Darwin, highlighted a poor state of reef development there and scattered occurrence (Darwin 1842; Dana 1975). Nowadays we know that region's circulation patterns, ENSO events, high environmental variability, and rapid bioerosion affect the suitability of conditions for reefs to occur (Dana 1975; Cortés 1997; Manzello et al. 2008). In 1992, Glynn & Colgan reported the widespread coral bleaching event of reef-building corals in Costa Rica, Panamá, Colombia and Ecuador after a 10-month sea warming period during the 1982-1983 ENSO. Manzello et al. (2008) found that low cement abundances and carbonate saturation state characterize ETP reefs, and suggested that these, along with elevated nutrients in upwelled waters, limit the cementation and stimulate bioerosion in ETP reefs.

Due to the extreme environmental conditions of the ETP, its coral reefs have low structural complexity and diversity (Cortés 1997). The small, sheltered embayments and offshore islands of the ETP (e.g. Galapágos, Gorgona, Cocos, Coiba) harbour rugose reefs, but coastal systems are generally characterised by rocky reef habitat. Shallow waters are dominated by the branching coral *Pocillopora damicornis*, with the massive coral *Porites lobata* common deeper (Guzmán & Cortés 1993). The environmental conditions found in the ETP have fostered a significant differentiation in the ecological and life history traits of many species, as some of the present fauna have their origins in the Central and Indo-Pacific (Glynn & Ault 2000). A good illustration of this is *Pocillopora damicornis*, which is a minor reef builder with brooding reproduction in the Indo-West Pacific, that in the ETP broadcast spawns and forms large mono-specific carpets over many hectares of reef (Combosch & Vollmer 2011).

The ETP is biogeographically important because of its extreme isolation from potential migratory pathways from the western Indo-Pacific region, and its separation from the Caribbean-Atlantic basin caused by the emergence of the isthmus of southern Central America around 3.1 million years ago (Dana 1975; Cortés 1997; Baums et al. 2012; Castillo-Cárdenas et al. 2014). Within the region, the Eastern Pacific Barrier, which is a stretch of deep water at 5000-8000km that separates the Central Pacific from the Eastern Pacific, represents a substantial impediment to marine species dispersal (Briggs 1961; Baums et al. 2012; Wood et al. 2014). Despite this, migration events through this vast barrier of open ocean were successful and much of the reef fauna of the ETP are derived from the central Indo-Pacific (Bowen et al. 2013). Glynn & Ault (2000) suggest that a mixture of vicariance and dispersion events have played complementary roles in shaping modern ETP fauna, as it is possible to find: i) Indo-Pacific migrants that managed to disperse long distances to colonise the east Pacific, ii) numerous endemics that have evolved recently, and iii) a few relict species with a close affinity to west Atlantic ancestors.

The ETP is considered as one the most isolated biogeographic regions of the world's oceans (Combosch & Vollmer 2011). Genetic patterns found in ETP corals demonstrate the effects that geographic isolation has had on these organisms (Baums et al. 2012; Wood et al. 2014). For example, the trans-Pacific coral *Porites lobata* presents strong genetic dissimilarities between populations from the Central and Eastern Pacific, which are attributed to the failure of trans-oceanic dispersal, given the inability of coral larvae to survive crossing the Eastern Pacific Barrier (Baums et al. 2012). Likewise, the abundant *Pocillopora damicornis,* exhibits limited gene flow over large and fine scales (i.e. sub-regions, populations, and individuals within reefs) (Combosch & Vollmer 2011).

1.1.3. Species diversity

The coast of the ETP is morphologically heterogeneous, offering diverse habitats for species to colonize. Its habitats include coasts containing estuarine lagoons, rainforest, mangroves swamps and mudflats, high cliffs with pocket beaches, rocky shores, and long sand beaches (Prahl 1989; Lacerda et al. 1993). It has been suggested that species richness in the region is vastly underestimated, and that its marine biodiversity, especially for Colombia and Ecuador, is the least well documented in South America as a whole (Miloslavich et al. 2011). A recent review by Miloslavich et al. (2011) reports 6,714 species from four Protist groups (Dinoflagelata, Foraminifera, Radiolaria, Tintinnida), two plant phyla (algae, angiospermae), and 30 animal phyla. No species have been recorded from the Placozoa, Gnathostomulida, Micrognathozoa, Locilifera, or Nematomorpha but this is likely due to lack of taxonomic study rather than any real absence (Miloslavich et al. 2011). The ETP exhibits some of the sea's highest levels of endemism with 72% of the 1,222 fish species there described as endemic (Zapata & Ross Robertson 2006; Robertson & Allen 2008).

1.1.4. Natural resource productivity

The productivity of the ETP is almost unparalleled (Ryther 1969; Fiedler et al. 1991), and this is because of the region's physicochemical characteristics. The Pacific basin is the world's largest, deepest and oldest, with its rocks dated at 200 million years old (NOAA, 2012). It contains more than half of the free water on Earth, and all the world's continents would fit into its surface area of 155 million km^2 (NOAA, 2012). Deep water in this ocean has a very low oxygen concentration mainly because the water is so old and therefore lacks the ventilation that takes place in the surface mixed layer, where oxygen is supplied by the air-sea exchange (Reid & Mantyla 1978). The ETP contains a very large area of low oxygen called an *oxygen minimum zone* (OMZ), where dissolved oxygen concentration is below 20 $\mu mol\ kg^{-1}$ (Helly & Levin 2004; Franz et al. 2012)(Figure 1.3). Given their differences in density, the low oxygen waters of the ocean's subsurface are separated from the shallow-warm and nutrient-poor layers by a thermocline. The very frequent coastal upwelling of the ETP interrupts this stratified layer by bringing up cool and nutrient rich water from the bottom, while injecting new nutrients to the surface (Fiedler et al. 1991). This mixture of water masses generates a significant enhancement in primary production, generating a cascading effect in the trophic system. The organic matter enters the food chain to support blooms of phytoplankton and zooplankton which in turn support higher trophic levels and long food chains (Ryther 1969).

Wind patterns have important connections to coastal upwelling as they provide the local driving force for vertical water transport. Moreover, trade winds generate westward frictional drag on the ocean that results in an accumulation of warm water in the western Pacific that produces a small sea-level slope of ~0.5 m, and a greater thermocline slope of ~100 m, with the west side of the ocean remaining deeper and higher (Barber & Chávez 1986). As a result the east/west slope generates a basin-wide thermocline/nutricline slope that pushes the thermocline of the Eastern Pacific closer to the surface (Barber & Chávez 1986). This means that the band where oxygen levels are suitable for fish and other organisms to survive is narrow in the ETP, since very few species can inhabit OMZs (Levin et al. 1991). In low oxygen environments bacteria and meiofauna thrive at the expense of macro-fauna which is primarily controlled by oxygen (Levin et al. 1991; Stramma et al. 2012; Franz et al. 2012).

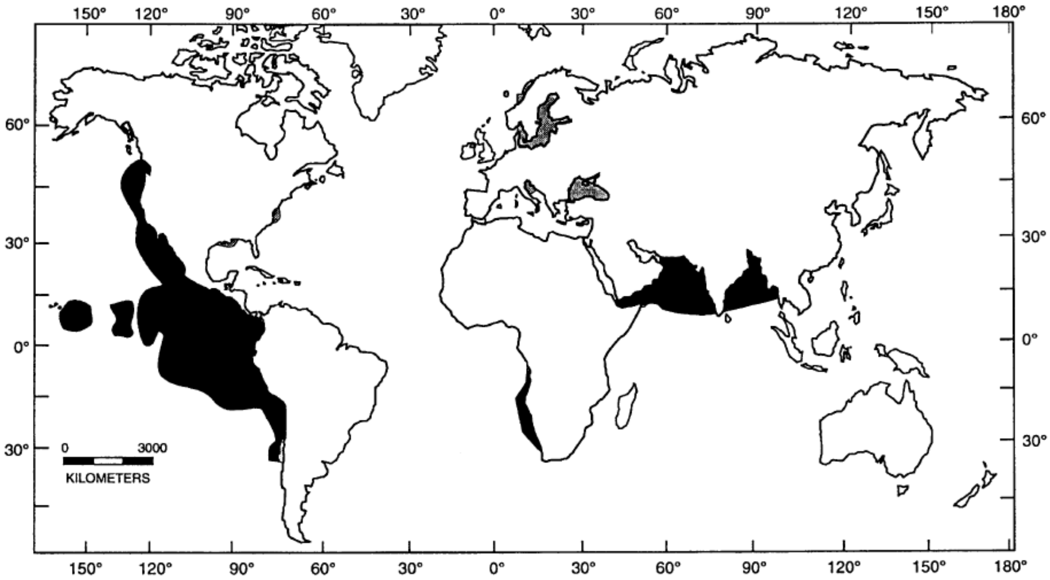

Figure 1.3. Location of the oxygen minimum zones throughout the oceans. Grey shaded areas represent hypoxic enclosed seas and fjords, and black shading depicts the open water oxygen minima. Taken from Levin (2003).

The boundaries of OMZs are influenced by currents, circulation, and climate regimes, therefore fluctuate with the sharp environmental dynamics of the ETP (Fuenzalida et al. 2009; Cepeda-Morales et al. 2013), such as El Niño events. OMZ fluctuations have been shown to impose considerable impacts on marine ecology (Helly & Levin 2004; Stramma et al. 2012; Gilly et al. 2013). OMZs are important oceanographic barriers that limit the movement and distribution of many species and therefore affect genetic diversity and evolution (Levin 2003). Since the 1960s low oxygen zones have increased to 4.5 million km^2, as result of high atmospheric and oceanic temperatures associated with global warming, and further declines in oceanic dissolved oxygen are predicted globally (Stramma et al. 2010).

While hypoxia tolerance can vary significantly between species and within stage-specific growth periods, the limitation of oxygen at any specific time can cause significant physiological problems for growth, reproduction success and survival in any species (Ekau et al. 2010). To survive hypoxic environments, animals have responded by adapting in ways which include: reduced size, high reproductive rate, adoption of dormancy stages, symbiosis with bacteria, lowered metabolic demands, utilizing anaerobic metabolism, enhanced surface to area in gills or branchiae, development of respiratory pigments, and behavioural adaptations like ontogenetic and vertical migrations (Levin 2003; Ekau et al. 2010; Seibel et al. 2016).

The horizontal and vertical expansion of OMZs has important consequences for the marine environment (Stramma et al. 2012; Gilly et al. 2013). Prince and Goodyear (2006) showed that lack of dissolved oxygen in the ETP restricts the vertical distribution of tunas, sailfish, and pelagic marlins by compressing their rich oxygen habitat into a very narrow surface layer of approximately 25 m. Stramma et al. (2012) calculated that the expansion of OMZ in the tropical northeast Atlantic caused a reduction of 15% in the vertical habitat of billfishes and tunas. This can make big pelagic species more vulnerable to overexploitation, because spending more time in the shallow thermocline increases the probability of them getting caught there by fishers (Prince & Goodyear 2006). As, high catch rates in habitat-compressed areas can provide false perceptions of thriving fish populations, fishery managers need to account for habitat compression in OMZ areas (Stramma et al. 2012).

For some species, the expansion of OMZs represents an opportunity to increase their distribution and abundance (Gilly et al. 2013). For example, the fast growing Humboldt squid (*Dosidicus gigas*) possesses physiological adaptations to thrive in OMZs, such as a metabolic suppression, which allows them to maintain very high levels of movement and foraging which contrast with other hypoxia tolerant cephalopods (Gilly et al. 2006; 2012). Zeidberg and Robison (2007) provide evidence that the range of Humboldt squid has expanded from the eastern equatorial Pacific to the North Pacific ocean since 1997. This large expansion has benefited Humboldt squid fisheries in the eastern Pacific (Gilly et al. 2013), but brings concerns about its effects on valuable commercial fish stocks, as the appearance of these opportunistic predators is correlated with declines of hake (Zeidberg & Robison 2007). Likewise, since jellyfish blooms are related to eutrophication and hypoxia, jellyfish are thought to increase in abundance with expanding OMZs, and can adversely affect fish (Purcell et al. 2007; Purcell 2012). As global warming creates conditions for hypoxia tolerant taxa to dominate, resulting biogeochemical and ecological changes in the ocean will have important effects on commercial fisheries especially in areas with strong upwelling like the ETP (Bakun et al. 2015).

1.1.5. Fisheries in the ETP

Fisheries contribute 6.3% of the gross domestic product in Ecuador, 3.9% in Colombia, 2% in Panamá, and 0.32% in Costa Rica (Boyd 2010), however this likely underestimates the significant contribution that the high productivity and marine biodiversity of the ETP (Ryther 1969; Miloslavich et al. 2011) makes to coastal livelihoods. The fisheries sector provides important food security and employment to countries in the ETP (MacKenzie 2001; Moreno-Sánchez & Maldonado 2013). Pelagic stocks, notably sardines, anchovy, and tuna represent the majority of catches, followed by demersal shrimp, lobster and crab (Díaz et al. 2011; Trujillo et al. 2012; Harper et al. 2014). A wide variety of fishing techniques and gears are used, ranging from industrial trawlers to low tech, non-mechanised artisanal canoes propelled by wind or paddle (Boyd 2010; Vega et al. 2014). Artisanal fisheries of the region are characterized by low-technology gears, such as gillnets, hand line, beach seine and cast nets (Trejos et al. 2008; Díaz et al. 2011). Collection of molluscs by hand from areas of mangrove is another important subsistence activity (MacKenzie 2001; Ocampo-Thomason 2006).

Worldwide, the livelihoods of more than 200 million people depend on small-scale fisheries to some extent (FAO 2008). Artisanal small-scale fisheries are highly valuable in their contribution to food security (McClanahan et al. 2015), local communities' livelihoods (Smith et al. 2005; Béné 2009), and the local economy (Smith et al. 2005). In developing countries fishing communities are marginalised, and if this is coupled with isolation from infrastructure and access to alternative livelihoods then reliance on fishery resources is further increased (Maldonado & Moreno-Sánchez 2014; Tilley & López-Angarita 2016). If high dependence on natural resources leads to high levels of overfishing then at worst this can threaten ecosystem health and viability in general (Cinner & McClanahan 2006).

In recent years, many ETP fisheries have shown catch declines as fishing effort has increased (Medina et al. 2007; Guzmán et al. 2008; Harper et al. 2014), suggesting a breach of maximum sustainable yield (Díaz et al. 2011). In Panamá, the snapper fishery collapsed in the Gulf of Montijo because of excessive use of gillnets (Mate 2005). Similarly, the once important scallop fishery in Panamá's Las Perlas archipelago collapsed in 1991, and by two decades later had shown no sign of recovery (Medina et al. 2007). In Colombia, there has been a clear reduction in abundance and size of the cockle *Andara tuberculosa*, which is the principal mollusc extracted by artisanal fisheries in the ETP (MacKenzie 2001). Similarly in Colombia, overexploitation and bad management of white shrimp (*Litopenaeus occidentalis*) and titi shrimp (*Xiphopenaeus riveti*) led to the decline of these fisheries (Díaz et al. 2011).

To mitigate fishery declines, managers can implement input controls such as restricting the amount of fisher/boat licenses, or output controls, like limiting the total catch (Purcell & Pomeroy 2015). However, both these approaches are challenging in the ETP, because limited data, poor management capacity, and weak legislative infrastructure (Mate 2005; Trejos et al. 2007; Fundación Futuro Latinoamericano 2011) lead to frequent exclusion of important socioecological factors that directly influence management success. In the absence of data with which to precisely calculate fish stocks and quotas, marine reserves can be introduced to provide refuges from fishing pressure to foster recovery of fished populations (Gell & Roberts 2003). However, the efficacy of marine reserves is highly affected by: reserve placement, size, spacing, and connectivity of habitats (Roberts et al. 2003; Gaines et al. 2010) and their capacity to increase fish stocks appears most effective in heavily overfished areas (Buxton et al. 2014). Placement of marine reserves in key habitat such as a nursery ground, can help replenish fisheries and is known to benefit conservation (McNally et al. 2011). In tropical coastal ecosystems, mangroves are an important nursery area for marine species (Mumby et al. 2004) and have been shown to enhance fishery yields (Aburto-Oropeza et al. 2008). Effective identification and protection of nursery areas can facilitate the multiple successes of safeguarding economically important juveniles and habitat, thereby fostering biodiversity, and helping support local livelihoods.

1.1.6. Mangroves in the ETP

Largely restricted to tropical and subtropical latitudes, mangroves are the only forest that can live at the confluence of land, freshwater and sea (Hogarth 2007). A global figure for total mangrove area in the year 2000 was estimated at 137,760 km^2 by Giri et al. (2011). Recent estimates using different techniques have calculated a total of

81,484 km^2 for 2014 (Hamilton & Casey 2016). In this case it is recognized that the difference in area between the years does not reflect mangrove loss but merely the approach used to calculate area and the definition of mangrove used in the two analyses (Hamilton & Casey 2016).

Two major floral realms are widely recognized in mangrove distribution: the Indo West Pacific (IWP) and the Atlantic East Pacific (AEP) (Hogarth 2007; Lo et al. 2014). The IWP extends from East Africa eastwards to the Central Pacific, comprises 57% of global mangrove area (Spalding et al. 2010), and is rich in species with 58 from 23 genera (Duke et al. 1998). The AEP encompasses all of the Americas, plus West and Central Africa and contains 13 species of 8 genera (Duke et al. 1998) in its 43% of the global mangrove area (Spalding et al. 2010). The major differences in floral composition of mangroves in tropical America started to develop after the closure of the Panamá isthmus 3.1 million years ago, which separated the Pacific from the Atlantic (Castillo-Cárdenas et al. 2014). After this, very particular climatic processes that started in the Miocene, shaped the actual mangrove flora to separate this into species restricted to seasonal dry climates, and species restricted to high precipitation (Jimenez 1999). The largest areas of mangroves in South America occur in the humid coastline of the Colombian Pacific and in the north of Ecuador (Esmeraldas region), whereas dryer areas in southern Ecuador have limited mangrove cover (Spalding et al. 2010).

Mangroves in the ETP have relatively low diversity, with only 11 species, with the most common genera of mangroves being *Rhizophora* and *Avicennia* (Yañez-Arancibia & Lara Domínguez 1999). As a general pattern for ETP countries, mangroves are abundant on the Pacific coastline as many estuaries, bays and rivers provide a suitable environment for their development and extensive coverage (Figure 1.4), contrasting with the sandy and high-energy shoreline of the Atlantic (Spalding et al. 2010). This topographic complexity means that the Pacific coast of Costa Rica is about 5 times the length of its Caribbean coast, and as such it harbours 99% of the country's 412 km^{2}total mangrove area (Silva-Benavides 2009). A high number of tidal flats and the significant freshwater input from upstream allow mangrove trees of the Pacific coast to grow up to 30 m in height, while in the Caribbean trees do not exceed 5 m (FAO 2007a). Dominant species are *Rhizophora mangle, Pelliciera rhizophorae,* and *Rhizophora racemosa* (Silva-Benavides 2009). One of the biggest areas of mangrove, the Golfo de Nicoya, has strong climatic contrasts where external parts of the Gulf receive yearly precipitation of around 2000-3400mm and have very productive forests with basal areas of 30 m^2/ha, while the inner part of the Gulf receives ~1600mm and has a less productive forest with a smaller basal area (20m^2/Ha) (Jimenez 1999). The most structurally complex forests occur in the Térraba-Sierpe wetland, where *R. racemosa, R. mangle* and *A. germinans* can reach heights of up to 40m (Silva-Benavides 2009).

Following regional trends, mangroves of Panamá are more abundant and form the principal forest in brackish areas of the Pacific coast, with the red mangroves (*R. mangle* and *R. harrisonii*) being the dominant species (FAO 2007b). In the Gulfs of San Miguel and Chiriquí, dense canopies of *Rhizophora* forests reach 30-40 meters (FAO 2007b). Precipitation over the country ranges between 1000 and 7000 mm per year, with 60% of the total river runoff of the Isthmus of Panamá draining towards the Pacific coast, where the tidal amplitude is wide (D'Croz 1993). Panamá Bay is known to hold 20% of Panamá's total mangrove area and has been designated as Wetland of International Importance, an Important Bird and Biodiversity Area and a Western Hemisphere Shorebird Reserve (Castellanos-Galindo et al. 2017).

In Colombia, mangrove forests are distributed from the frontier with Ecuador to Cabo Corrientes as a thick parallel band all along the Pacific coast, occasionally penetrating more than 20 km into the continent to form a lush forest (Prahl 1989). The rich flux of rivers and wide continental platform of soft soil provides the ideal productive environment for their success (Prahl et al. 1990). From Cabo Corrientes up to the frontier with Panamá, mangrove forests become patchy, as few big rivers wash through these lands and the rocky escarpment coast can only support species able to colonize hard soils such as *Pelliciera rhizophorae* (Ministerio de Medio Ambiente 2002). The Colombian Pacific has the highest precipitation on the planet, allowing for very rich mangrove areas where trees can reach up to 40 m in height and host many species of birds, mammals, reptiles, fish, crabs and molluscs, including the "piangua" *Anadara spp*., a bivalve that supports a very important local artisanal fishery (Delgado et al. 2010).

Fringing mangroves extend throughout the steep coast of Ecuador and Galápagos Islands, whereas broader formations can be found around estuaries and river basins.

The biggest forests are located in the estuary of the Guayas River, in the Gulf of Guayaquil which is the largest estuarine ecosystem of the Pacific coast of South America, and in the high precipitation zones around the estuary of Santiago-Capas-Mataje, where mangrove trees reach their pinnacle of development growing up to 50 m in height (Twilley et al. 2001; FAO 2007c).

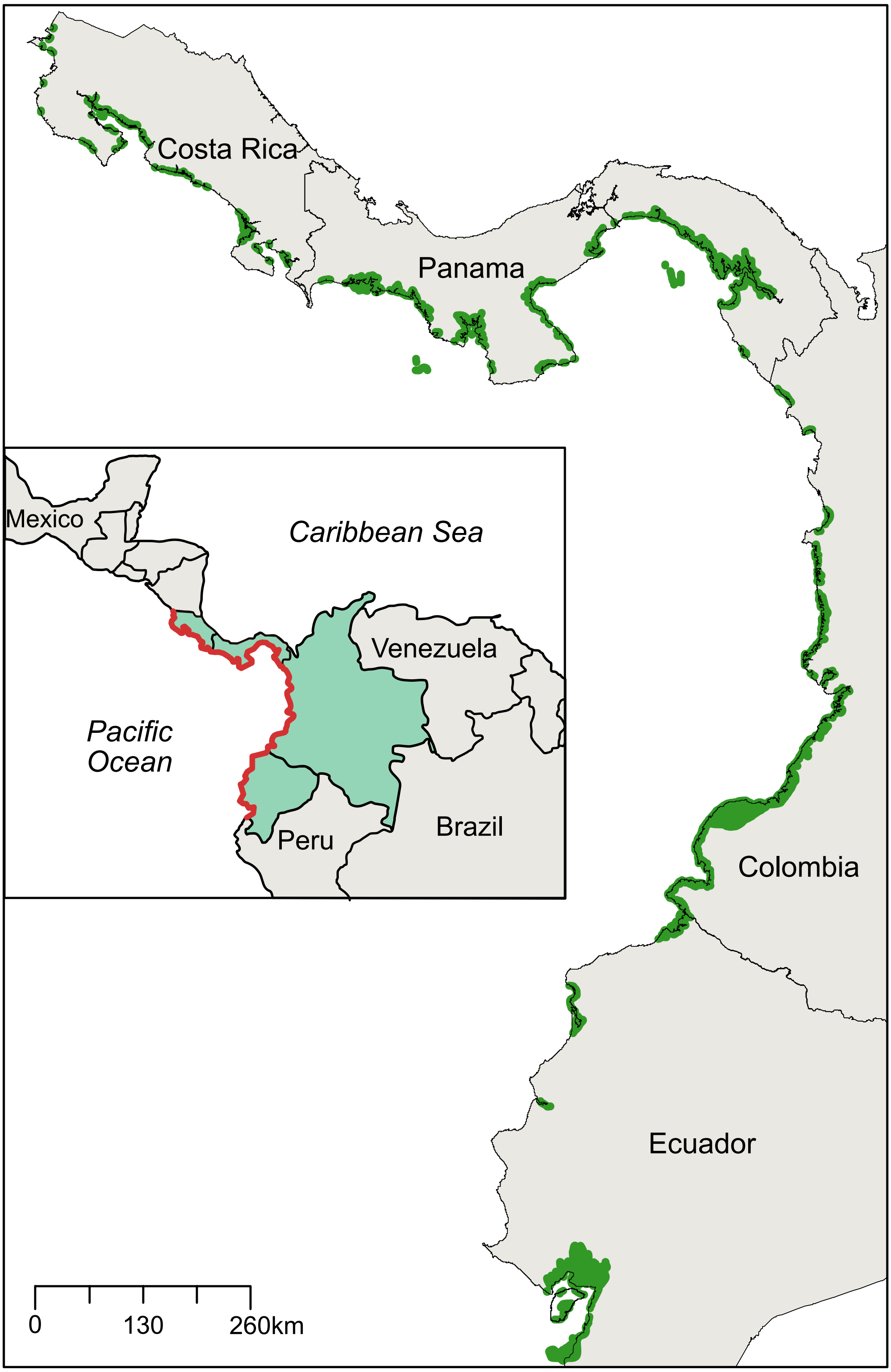

Figure 1.4. The distribution of mangroves in the Eastern Tropical Pacific region according to Giri et al. (2011).

Mangroves in the ETP were utilized by pre-Columbian human settlements around 5000 years ago for timber, charcoal, tannins, and fishing (Prahl 1989; Prahl et al. 1990; D'Croz 1993). In the first half of the 20th century, wood exploitation was common for tannin extraction and big trunks were used for house construction and building electric posts and railway tracks (Prahl 1989). In the second half of the 20th century further product uses were developed for things like charcoal production and mangrove land itself was deliberately converted to use for agriculture and aquaculture, with this being rapidly developed after 1969 (Ocampo-Thomason 2006). In 30 years, 57% of mangrove area was lost from Ecuador (Ocampo-Thomason 2006), though more conservative estimates put it at 20-30% (Shervette et al. 2007). As a general trend, most mangrove deforestation occurred after 1950s (Valiela et al. 2001; Alongi 2002; Duke et al. 2007; López Angarita et al. 2016) but recent research points at the reduction of global deforestation rates (Hamilton & Casey 2016). For a more complete account of the historical trends of mangrove decline in the ETP see Chapter 2, and for recent deforestation trends Chapter 5.

1.1.7. Ecosystem services provided by mangroves

Mangroves are an ecosystem of high structural complexity and diversity that provide multiple ecosystem services. They play an important role in worldwide primary productivity, organic matter storage, coastal protection, and conservation of biodiversity (FAO 2007a). However, high rates of mangrove loss have been recorded globally (Valiela et al. 2001; Alongi 2002; Duke et al. 2007). To effectively promote mangrove protection it is key to understand the importance of mangroves for society. Ecosystem services represent the benefits to humans provided by nature and their economic valuation helps decision makers appreciate the worth of a healthy environment to society, and therefore the importance of taking conservation measures to help achieve this (Laurans et al. 2013; Mukherjee et al. 2014). In particular, mangroves offer myriad ecosystem services and are essential for the survival of many communities in tropical developing countries (Walters et al. 2008; Hussain & Badola 2010).

Below, I review the ecosystem services provided by mangroves following a structure outlined by The Economics of Ecosystem and Biodiversity initiative (www.teebweb.org).

- *Provisioning services: the material and/or energy outputs from mangrove forests*

Mangroves support artisanal livelihoods and income for the millions of people who live beside them by providing forest products such as wood, tannins and honey (Bandaranayake 1998; Tallis et al. 2008).

- *Food:* this includes edible fruits (e.g. *Sonneratia caseolaris*), flower buds (e.g. *Rhizophora apiculata*), and propagules (e.g. *Bruguiera cylindrical*, *B. gymnorhiza,* and *B. sexagula*) that communities collect and eat raw, cooked, and in juices (Baba et al. 2013). Honey can also be harvested from mangrove trees, which is of particular importance in the Sundarbans in Bangladesh (Baba et al. 2013). Finally, the great diversity of animals that inhabit mangroves (e.g. shrimp, crabs, cockles and fish) represent an important source of protein and income for coastal communities.
- *Raw materials:* The strength and durability of mangrove wood made it historically sought after for multiples uses in construction, and for charcoal and tannin production (Walters et al. 2008). Some communities rely on mangrove wood as the major source for house construction, fuel wood, charcoal, and boat building (Dahdouh-Guebas et al. 2000; Baba et al. 2013).

- *Regulating services: the regulatory role provided by mangroves*

- *Carbon sequestration and storage:* Mangroves are known to capture and accumulate carbon at a higher rate than terrestrial forests, which makes them the most carbon rich forests in the tropics, containing on average 1,023 Mg C per hectare (Donato et al. 2011). Their ability in this respect has recently been the focus of much research and is highlighted by Alongi (2012), Pendleton et al. (2012), Bianchi et al. (2013), Lee et al. (2014), Adame et al. (2015), Zarate-Barrera & Maldonado (2015), Sahu et al. (2016), Alongi et al. (2016), and Marchio et al. (2016) amongst others. Given their significant role in reducing CO_2 concentration in the atmosphere, the importance of conserving mangroves for climate change adaptation and mitigation has been recognized (Siikamäki et al. 2012; Duarte et al. 2013; Murdiyarso et al. 2015). It has been calculated that the 35% global loss of mangrove forests during the decades between 1980-2000, represents a release of more than 3.8×10^8 tonnes of carbon previously stored as mangrove plant biomass (Cebrian 2002). However, deforestation does not only involve the loss of carbon stored in standing biomass, but also the efficiency to which mangroves store carbon in adjacent sediments (Bouillon et al. 2009). Granek & Ruttenberg (2008) compared mangrove intact plots with cleared plots deforested 8 years before in the Panamánian Caribbean, and showed that sediments of intact plots have 50% more organic matter than cleared plots. Moreover, Irving et al. (2011) demonstrated that extensive mangrove loss since the 1970s has decreased their average global rate of carbon capture and storage, from ~26.5 x 10^6 t C yr^{-1} to a current average of ~17.3 x 10^6 t C yr^{-1}.

- *Moderation of extreme events:* Coastal populations are highly vulnerable to extreme events, such as storms, cyclones and hurricanes. Mangrove structure and their complex root systems offer protection to coastal settlements against floods, tropical storms, sea level rise, and solar UV-B radiation (UNEP-WCMC 2006). After the occurrence of an extreme event, dense vegetation can reduce water flow velocities, turbulent flows, gusting winds and wave swell (Spalding et al. 2014). It has been shown that mangroves play an important role in storm surge reduction by slowing the flow of water, decreasing surface waves more than 75% over 1 km of mangroves (McIvor et al. 2012b), and reducing wave height by between 13% and 66% over 100 m of mangroves (McIvor et al. 2012a). During storms, hurricanes and periods of high wind, mangroves can reduce wind and wave swell helping to avoid flooding and infrastructure damage (McIvor et al. 2012a; Das & Crépin 2013). Moreover, research shows that mature mangrove forests can help to mitigate damage from tsunamis, as larger trees with a stem diameter of 25–30 cm are more successful in surviving tsunami impacts than those whose diameter is 15–20 cm (Yanagisawa et al. 2009). Dahdough-Guebas et al., (2005) interviewed local inhabitants of areas hit by tsunamis and concluded that mangroves played a critical role in storm protection, but that the extent of protection depended on the quality of the forest. It has been suggested that mangrove area is a predictor of the degree of natural coastal protection, with larger mangrove area related to less human deaths after cyclones (Das & Vincent 2009).
- *Waste water treatment:* Domestic sewage and aquaculture wastewater commonly wash the coasts of developing countries. Mangroves are highly efficient water filters (Kim et al. 2016) and can decrease coastal pollution by

acting as nutrient traps (Bouillon et al. 2009), hence they act as natural water treatment processers. They are able to remove contaminants through tidal flushing, mangrove plant assimilation and sediment microbial metabolism (Ouyang & Guo 2016). In particular, the ability of both mature trees and seedlings to remove eutrophic pollutants and suspended solids from aquaculture wastewater has been highlighted (Gautier et al. 2001; De-León-Herrera et al. 2015; Ouyang & Guo 2016). Therefore, mangrove-aquaculture coupling systems have been promoted as way forward to reduce the negative impacts of aquaculture on the environment (Primavera 2006). Furthermore, mangroves can also filtrate anthropogenic nutrients by acting as a natural buffer between upstream wastewater sources and the sea (Lin & Dushoff 2004; Zhang et al. 2010). It has been shown that mangroves located at a closer distance to urban centres hold higher levels of metal pollution compared to those in rural areas (Mremi & Machiwa 2003). Mangrove sediments absorb and hold heavy metal pollutants (Almahasheer et al. 2014), preventing their spread to other coastal habitats (Lacerda 1998).

- *Erosion prevention:* Mangroves are active and integral agents contributing to vertical accretion by enhancing sedimentation and directly adding organic matter to the soil (Lee et al. 2014). Mangroves drive the increase in soil volume by accelerated sedimentation, sediment trapping, and organic matter input (Lee et al. 2014). McKee (2011) experimentally demonstrated the above and below ground contribution of mangrove root production and benthic mat formation to accretion and elevation gain. The study of mangroves' role in sediment accretion and vertical land development allows coastal land managers to understand how this ecosystem adjusts to sea level rise (Krauss et al. 2013).
- *Maintenance of soil fertility:* Mangroves naturally occur in low nutrient environments and in response have evolved traits to acquire and conserve nutrients, such as evergreeness, resorption of nutrients, and high root/shoot ratios (Reef et al. 2010). Mangroves act as nutrient traps by recycling key nutrients such as carbon, nitrogen and sulphur (Kathiresan & Bingham 2001). The organic content of sediment has been shown to be significantly higher in intact than in cleared mangroves (Granek & Ruttenberg 2008). Hussain and Badola (2008) analysed nutrient contents in mangrove and non-mangrove soils around India's Bhitarkanika National Park and found that mangrove soil

contained several nutrients including nitrogen, phosphates and potassium that enhanced the production of adjacent agroecosystems.

- *Supporting services of mangroves*

- *Habitats for species:* Mangrove structural complexity and high productivity provide refuge and food for critical life stages of many commercially important species of fish and shellfish (Nagelkerken et al. 2001; Aburto-Oropeza et al. 2008). For example, many marine species use mangroves as an intermediary nursery habitat before migrating as adults to reefs or offshore (Mumby et al. 2004). When Nagelkerken et al. (2002) compared the densities of 17 Caribbean reef species between reefs of islands with and without mangroves and seagrass beds they found that 11 species, many of them commercially important, were absent or had low densities on islands lacking mangroves. Moreover, in the Zancudo forest of Golfo Dulce, Costa Rica, an inventory of fish biodiversity showed that more than half of the 82 fish species found in mangroves were of importance for artisanal fishers, with only 11 reported to also occur in other habitats (Feutry et al. 2010). However, the nursery value of mangroves varies according to their spatial extent and temporal accessibility, and these in turn differ greatly between and within biogeographic regions (Lee et al. 2014). For this reason, the significance of mangroves as nursery sites remains controversial (see section 1.1.8). Liquete et al. (2016), suggest that the nursery function should be considered as an ecosystem service in its own right when it is linked to a concrete human benefit such as increased fishery yields

- *Cultural services of mangroves which include recreation facilities for the benefit of mental and physical health; tourism; opportunities for aesthetic appreciation and inspiration for culture, art and design; spiritual experience and sense of place*

Evidence indicates that experiencing nature and biodiversity is linked to improved human health and well-being (Sandifer et al. 2015). Historically, mangroves have provided local communities with aesthetic experiences, spiritual enrichment and cultural inspiration (Rönnbäck et al. 2007; Walters et al. 2008; Uddin et al. 2013; James et al. 2013). Some communities, have been using mangroves for centuries and they have become a fundamental part of their livelihoods, culture and identity (UNEP 2014). The spiritual and cultural value of mangroves is difficult to translate into monetary terms, but their recreational value has been quantified for tourism. Uddin et al. (2013) performed an economic valuation of ecosystem services in the world's largest mangrove system, the Sundarbans Reserve Forest in Bangladesh, and estimated their contribution via tourism to the local economy was US$42,000 per year. Economic valuation assessments have shown that ecotourism in mangroves has high economic potential, which should be developed using the appropriate incentives and in collaboration with local communities (Mukherjee et al. 2014). One of the most visited mangrove areas in the ETP is Térraba-Sierpe in Costa Rica, which holds around 40% of the total mangrove forest of the country and is considered to be one the best preserved forests (Silva-Benavides 2009).

1.1.8. Mangroves and fisheries

Mangroves provide a suitable habitat for a very diverse array of marine and intertidal species, which can be permanent residents or temporal visitors arriving with the flooding tide (Dorenbosch et al. 2006; Hogarth 2007; Duke 2011). Mangroves are hypothesized to be nursery areas because they offer refuge, food, protection from predation, and critical life stages for many fish/shellfish species, thanks to their complex physical structure, shallow water microhabitats and reduced visibility (Beck et al. 2001; Lee 2008; Nagelkerken et al. 2008; Lee et al. 2014). According to Beck et al. (2001) an area counts as nursery habitat if its contribution per unit area to the production of individuals that recruit to adult populations, is greater than production from other habitats in which juveniles occur. However, only a few studies that have evaluated what species actually use mangroves as nursery grounds (Sheaves 2017), and mangrove function in this respect is still under debate (Nagelkerken et al. 2008; Lee et al. 2014). Moreover, it has been argued that the widely held assumption that 75% of commercially caught fish depend directly on mangroves, is illogical and inaccurate, and cannot be traced back to scientific data (Sheaves 2017).

In attempting to answer the question of how important mangroves are for fish, many have explored the link between fisheries productivity and mangrove extent, finding positive statistical relationships between catches of fish/shellfish and mangrove area (Pauly & Ingles 1986; Aburto-Oropeza et al. 2008; Turner 2011; Carrasquilla-Henao & Juanes 2016). For example, Aburto-Oropeza et al. (2008) found landings of mangrove-related fish and crab species were positively related to the productive area in the mangrove-water fringe. However, given that correlation does not necessarily imply causality, a number of studies that have linked fisheries productivity to mangroves have been found to have analytical problems such as: (i) temporal and spatial variability, (ii) differences of scale, (iii) few predictor variables, and (iv) auto-correlation that makes it unclear as to whether mangrove presence is the causal factor or another of the many factors related to mangrove cover such as extensive shallow seas, intertidal area, tidal creeks, organic matter, or length of coastline (Baran & Hambrey 1998; Manson et al. 2005; Blaber 2007; Saenger & Funge-Smith 2013; Lee et al. 2014; Sheaves 2017).

Core to understanding the role of mangroves as nurseries and as such enhancers of fishery yields, is appreciation of the variation in the degree to which different species of fish/shellfish depend on mangroves (Blaber et al. 2000; Saenger & Funge-Smith 2013). In reality it is very difficult to classify the dependence of a particular species on a specific habitat because the degree of dependence can vary between/within species and life stages (Saenger & Funge-Smith 2013; Sheaves 2017). Studies that provide evidence on the nursery role of mangroves have also found high variability in the contribution of juvenile habitat to adult populations (Kimirei et al. 2013), indicating clear species-specific differences in nursery habitat dependency across seasons, years, or at different locations (Barletta et al. 2003; 2008; Kimirei et al. 2011; Castellanos-Galindo et al. 2013; Lacerda et al. 2014).The high spatio-temporal variability in ontogenetic habitat use shows the difficulty of generalizing nursery habitat definitions in heterogeneous habitats by fish species in different life stages (Dantas et al. 2011; Saenger & Funge-Smith 2013). Fish assemblages vary greatly between different estuaries, making it difficult to attribute changes in fish faunal composition to changes in the ecological condition of a habitat (Sheaves et al. 2012; Sheaves 2016). Additionally, environmental factors are bound to add another layer of complexity, as catchment hydrology, configuration of estuarine mouth, substrate and mangrove area have been shown to affect fish species composition (Ley 2005). In a meta-analysis, Igulu el al. (2014) found tidal amplitude and water salinity to be major drivers of mangrove fish habitat irrespective of the biogeographic region.

When exploring the link between mangrove area change and fisheries production, the effect of fishing itself must be accounted for (Saenger & Funge-Smith 2013). However, fishery data for small-scale fisheries is often absent, usually unreliable and often impossible to verify and compare. This is because data collection systems such as catch surveys from selected gear types, vessels, and landing areas that are used to characterize small-scale fisheries in the developing world, fall short at doing so because of the dispersed and informal nature of the fisheries in question (Staples et al. 2004; Mills et al. 2011; Rubio-Cisneros et al. 2016; Tilley & López-Angarita 2016). Therefore, fisheries independent data could help overcome deficiencies of fisheries catch data, and allow us to distinguish between changes in catch due to fishing pressure, and changes in catch caused by habitat loss (Saenger & Funge-Smith 2013).

Many studies have explored the value of the nursery role of mangroves as an ecosystem service (Barbier & Strand 1998; Rönnbäck 1999; Sathirathai & Barbier 2001; Gunawardena & Rowan 2005; Aburto-Oropeza et al. 2008; Barbier et al. 2011; Brander et al. 2012; Uddin et al. 2013; Hutchison et al. 2014). Anneboina et al. (2017), used an econometric framework and estimated the marginal effect of mangroves to the production of marine fisheries in India as 1.86 tonnes per hectare per year, which represented a contribution of 23% to the marine fish output of India in 2011. Aburto-Oropeza (2008) estimated the annual value of the services provided to fisheries by mangroves in the Gulf of California at between US$25,000 and US$50,000 per hectare. Mukherjee et al. (2014) reviewed existing literature for monetary valuations of ecosystem services provided by mangroves and found that fisheries generated the highest mean economic value compared to other services. When valuing mangrove-associated fisheries, such values should not be seen as a one-dimensional economic statistic but as multidimensional, depending on which sector of society and context they relate to (Hutchison et al. 2014). For example, small-scale fisheries provide more value in terms of livelihoods and employment than high value shrimp aquaculture, and cockle fisheries are highly valuable for food security and income for the most vulnerable sectors of society (MacKenzie 2001; Hutchison et al. 2014). More accurate valuation requires a better understanding of the linkages between mangroves and fisheries. Saenger & Fungue-Smith (2013) recommend that future evaluations should include i) better account of the life cycles of commercial species, ii) identification and quantification of links between juvenile and adult populations including recruitment, iii) standardised sampling methods, and iv) fishery dependent and independent data.

1.2. Research Objectives

1.2.1. Justification of research

Given the documented importance of mangrove forests, research is needed to support conservation strategies to identify current pressing threats to mangroves, and to better characterize their role in supporting artisanal fisheries and associated livelihoods. Improved management of mangroves and fisheries is of great importance in the ETP given local communities' strong dependence on natural resources, low education level, and limited opportunity for income diversification (García 2010; Pinto & Yee 2011; Salas et al. 2012).

The aim of my research is to further our understanding of mangroves and artisanal fisheries in the ETP by i) evaluating mangrove condition at a regional level by mapping human activities that threaten mangroves; ii) using landings data from artisanal fisheries to explore the contribution of mangroves to fisheries and livelihoods; iii) providing a mangrove coverage and land use baseline for governments to shape policy based on historical decline trends; and iv) evaluating current protection effectiveness for mangroves in the ETP. My results will provide decision makers with a clear picture of the history of mangrove management in the ETP, and with the scientific information necessary to frame a sustainable management strategy for mangroves in the ETP.

1.2.2 Research questions and objectives

In **Chapter 2** I ask: i) What are the dynamics of the relationship between mangroves and people, and the patterns of mangrove degradation through history in the ETP region? ii) What are the historical trends of mangrove area change? iii) What is the state of mangrove protection in terms of legislation, and number and coverage of protected areas? This chapter is structured as a review article and was published as: *López Angarita J, Roberts CM, Tilley A, Hawkins JP, Cooke RG. 2016. Mangroves and people: Lessons from a history of use and abuse in four Latin American countries. Forest Ecology and Management 368:151–162.*

Chapter 3 is a case study of the Gulf of Montijo, which is a RAMSAR site in Panamá. In this I explore the following question: what is the relationship between mangrove condition and species composition in artisanal fisheries landings? My objectives were to i) identify the main human activities impacting mangroves using satellite images and use this to develop a model of cumulative anthropogenic impact on mangroves, and ii) explore the links between landings composition and mangrove condition.

Chapter 4 is a case study on the effects of area-based fisheries management in the artisanal fisheries of Northern Chocó, Colombia. This chapter asks the questions: i) Does territorial use rights in fisheries increase productivity of small-scale fisheries? ii) Does fishery management affect catch composition, particularly for mangrove dependent species? To answer these, I compare fishery landings information from 2010 to 2013 between adjacent areas where one allows territorial rights use and the other is open access.

In **Chapter 5** I ask i): Are mangroves in Colombia, Costa Rica, and Panamá still declining in the XXI century? ii) What are the main land use activities around mangroves in the Pacific coast of these countries, and iii) how effective are protected area networks at conserving mangrove cover? As part of this my objectives were to i) calculate deforestation rates of mangroves between 2000 and 2012, and ii) map land use adjacent to mangroves inside and outside protected areas. This chapter offers relevant information for mangrove conservation in the ETP as it assesses the efficacy of mangrove protection policies there to date.

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Preface

Mangroves are important habitats for coastal populations, particularly in Latin America, where many poor communities rely directly on mangroves goods and services. However, mangroves have declined globally and conservation action is needed to reduce deforestation rates. To develop effective protection for mangrove habitats it is important to reconstruct historical decline of mangroves, additionally, understanding their interaction with humans through history can provide valuable lessons to incorporate in conservation strategies.

In this Chapter, I use historical ecology to reconstruct temporal trends of mangrove decline in four countries of Latin America, and review protected area coverage and policies involving mangroves. My results show that a significant proportion of mangrove area has been lost in this region, and that more reliable estimates of mangrove cover are needed. However, historical negative attitudes towards mangroves changed when the links between mangroves and human wellbeing were established. I highlight the need for creating regional initiatives for mangrove conservation for Colombia, Costa Rica, Panamá and Ecuador.

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I declare that the work submitted is my own. Co-authors contributed to ideas and insight, and helped witting the manuscript.

Chapter 2. Mangroves and People: Lessons from a History of Use and Abuse in four Latin American Countries

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Abstract

From native pre-Columbian subsistence economies to the modern global economy, mangroves have played an important role providing goods and services to human societies for millennia. More than 90% of the world's mangroves are located in developing countries, where rates of destruction are increasing rapidly and on large scales. In order to design effective conservation strategies, it is critical to understand the natural dynamics and anthropogenic drivers of these coastal wetland habitats. We use retrospective techniques to reconstruct mangrove forest history in the Eastern Tropical Pacific. We examine available, present day estimates of mangrove area and evaluate the representation of mangroves in the protected area systems of Costa Rica, Panamá, Colombia and Ecuador, evaluating existing policies regarding mangroves. Archaeozoological evidence shows that mangroves were exploited for many thousands of years by pre-Columbian societies. Post-conquest deforestation prevailed during the next 400 years. Since 1990, despite increasingly positive attitudes towards mangroves and their inclusion in protected areas and conservation policies, mangrove cover has continued to decline due to expanding human activities (agriculture, aquaculture, coastal development), even in the presence of laws prohibiting their removal. Here we provide an historical ecology baseline of mangroves in the Eastern Tropical Pacific, from which to view current trends and map future trajectories. Given the myriad negative consequences of mangrove loss recorded worldwide, and the strong ecological connectivity of the region, developing

effective strategies for mangrove management at an appropriate scale will be paramount to protect coastal livelihoods and biodiversity.

Key words: Archaeozoology, Protected area, Conservation, Management, Eastern Tropical Pacific, Historical ecology.

2.1. Introduction

Largely restricted to tropical and subtropical latitudes, mangroves are the only vascular flowering trees that can live in the confluence of land, freshwater, and ocean (Hogarth 2007). This involves adapting to fluctuating environmental conditions such as changes in salinity, regular soil inundation, shifting sediments, and in-water low oxygen concentrations (Kathiresan & Bingham 2001). As such, mangroves display a large set of morphological and ecophysiological adaptations to help them survive in these dynamic habitats. Among these adaptations are 1) the exclusion of salt by roots, 2) rapid canopy growth, 3) viviparous embryos, 4) tidally dispersed propagules, 5) exposed roots that breathe above ground, 6) highly vascularized wood, 7) efficient nutrient retention, and 8) salt-excreting leaves (Alongi 2002; Duke 2011).

Despite being considered a rare forest type because of their small global extent (less than 1% of tropical and subtropical forests worldwide), mangroves provide a wide range of ecosystem services and direct uses including coastal protection, fuel (charcoal, firewood), food (fruit, leaves, associated vertebrates and invertebrates), and construction material (Hogarth 2007; Brander et al. 2012). Even though the ecological importance of mangroves has come to be widely recognized, reports of the Food and Agriculture Organization show a widespread decline of mangrove area, with losses over 20% of total global coverage (36,000 km^2) between 1980 and 2005 alone (FAO 2007). It has been claimed that estimated rates of mangrove loss are three to five times greater than the overall loss rates calculated for other forests and coral reefs on a global scale (Valiela et al. 2001). In light of the current degradation rate of marine ecosystems, intensifying anthropogenic impacts, and climate change, protected areas emerge as an essential strategy for conservation. More than 90% of the world's mangroves are located in developing countries, where rates of destruction are rapidly increasing (Duke et al. 2007; Webb et al. 2014; Richards & Friess 2015).

In order to design more effective conservation strategies, it is critical to understand long-term anthropogenic effects as well as the natural dynamics of this marine-coastal habitat in space and time. Study of forests' past dynamics represents a fundamental insight (Dahdouh-Guebas & Koedam 2008). Pre-Columbian societies represent more than 95% of the history of mangrove-human interaction in the neotropics. Historical ecology techniques derived from social sciences, such as exploration of documentary archives and archaeological investigations, can provide valuable information and complement other direct survey methods (e.g. remote sensing) applied to understand current processes and map possible future trajectories (Cormier-Salem 1999; Dahdouh-Guebas & Koedam 2008).

The Eastern Tropical Pacific (ETP), is a distinct marine ecoregion encompassing continental shore between southern Baja California to northern Perú including oceanic island groups such as Cocos, Malpelo, and Galápagos oceanic islands (Spalding et al. 2007; Briggs & Bowen 2012). However, within this broader region, the Pacific waters of Panamá, Costa Rica, Colombia and Ecuador (Figure 2.1) share particular conservation significance by containing a cluster of World Heritage Sites (Edgar et al. 2011). This area, termed in this study as the ETP, is an important biogeographical region in terms of marine resource productivity and biodiversity, supporting a range of rich fisheries and exhibiting many endemic species (Zapata & Ross Robertson 2006; Fiedler & Talley 2006). In this paper we use retrospective techniques to reconstruct mangrove forest history in ETP countries, with the aim of understanding the historical reasons behind mangrove deforestation in the region. Additionally, we review available estimates of mangrove area from recent years and evaluate the representation of mangroves in the marine protected areas of four countries, examining existing policies regarding mangrove conservation. The historical timeline of ETP mangrove forests and their current protection status that we present improves our understanding of the relationship ~~dynamics~~ between mangroves and humans, and provides a regional information baseline from which governments can build improved management strategies.

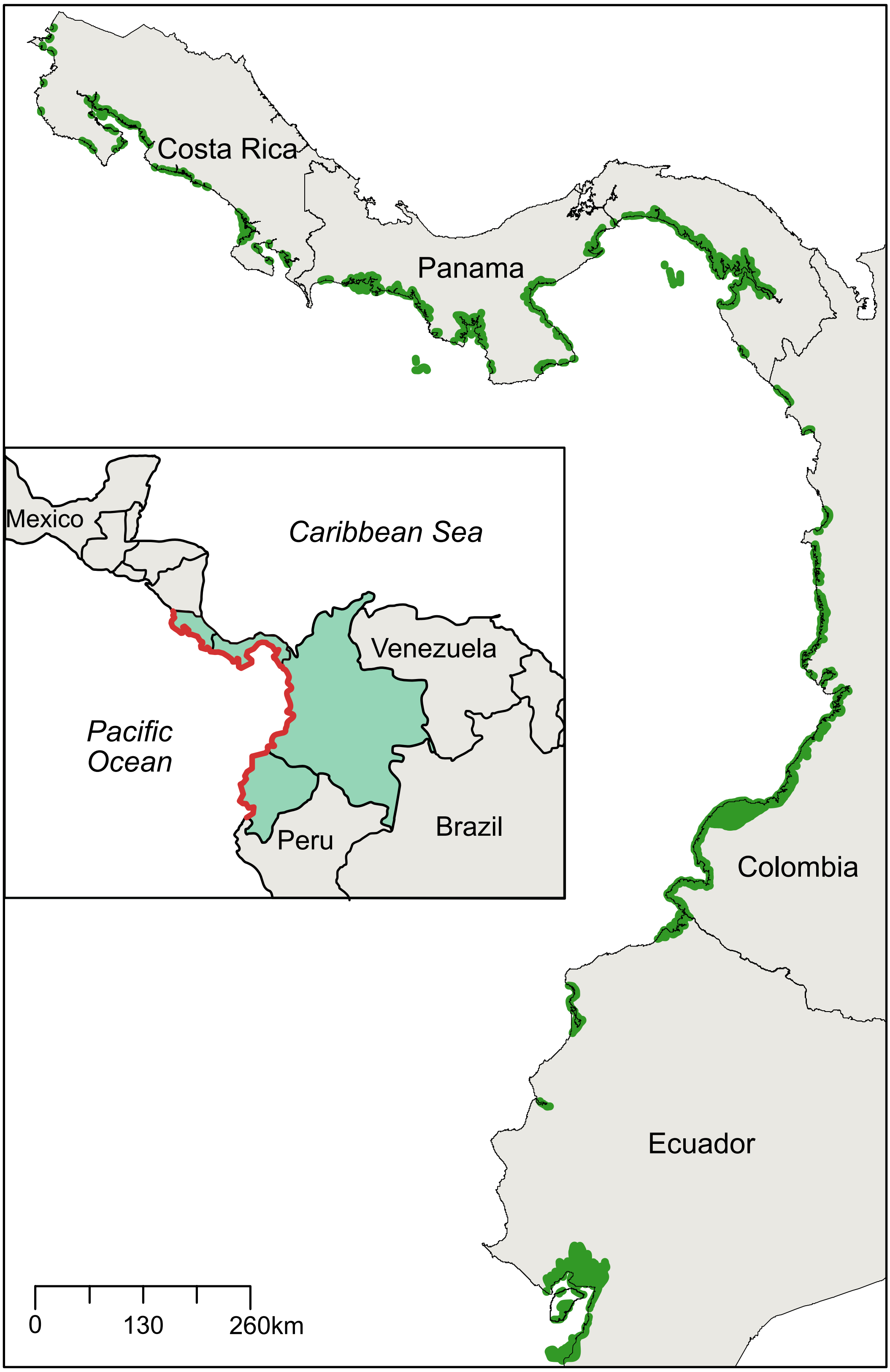

Figure. 2.1. The Eastern Tropical Pacific region encompasses the Pacific coasts of Panamá, Costa Rica, Colombia and Ecuador (red line, inset map). The distribution of mangroves in the region according to Giri et al. (2011) is shown.

2.2. An historical timeline of mangrove decline

2.2.1. Evolution of perceptions & attitudes towards mangroves

Mangroves' ability to thrive in salt water has attracted substantial scientific attention and academic curiosity, especially among botanists (Lugo & Snedaker 1974; Saenger et al. 1983; Walters et al. 2008). However, since our understanding of the services provided by mangroves as a coastal habitat have been scant and defective until recently, attitudes towards mangroves as an ecosystem have been ambivalent (Lugo & Snedaker 1974).

As far as we know, the first descriptions of mangroves by ancient literate observers were made in the year 325 BCE by Nearchus, the Greek Admiral of Alexander the Great's fleet. In the '*Chronicles of Nearchus*' he described mangroves in the Red Sea, the Persian Gulf, and the Indus Delta (Bowman 1917). Twenty years later in 305 BCE Theophrastus, a pupil of Aristotle, also referred to mangroves in his '*Enquiry into Plants*' (Schneider 2011): "*But there are plants in the sea, which they call 'bay' and 'olive' […]. On the islands which get covered by the tide they say that great trees grow, as big as planes or the tallest poplars...*" (Hort 1916). On the American continent, the first Spanish chronicler to describe mangroves from a botanical standpoint was Gonzalo Fernández de Oviedo in his '*General and Natural History of the Indies*' in 1531, placing emphasis on their usage by indigenous communities: *"Mangrove is one of the best trees in these lands, and it is common in these islands (Greater Antilles) and in Tierra Firme (mostly the Isthmus of Panamá). Its wood is one of the best ones around for building shelves, poles, posts for houses, fences, window frames, and doors and other small things…the bark of these mangroves is singularly good for tanning cow leather in a short time"* (de Oviedo y Valdés 1535).

Literature about mangroves between the 17th and 20th centuries focused mainly on describing mangrove morphology, habitat, distribution, species diversity, taxonomy, and systematics (Bowman 1917). From around 1900 onwards, studies that highlighted the ecological role of mangroves (mostly regarding their functions of sediment consolidation and shoreline maintenance) emerged to join earlier descriptive literature (Lugo & Snedaker 1974). The second half of the 20th century signalled the first public initiatives for mangrove conservation following work that highlighted their economic value for Florida fisheries due to their role in food web enrichment (Lugo & Snedaker 1974). Thus, the history of society's perceptions towards mangrove systems has evolved only recently from being considered as a barren wasteland of unhealthy soils, to being complex ecosystems upon which humans depend. The following sections explore the decline of mangrove forests in the ETP region.

2.2.2. Early pre-Columbian societies

As the 16th century quote from Oviedo (above) implies, mangroves played a crucial role in the way of life for many coastal societies, and are closely linked with human culture. In India, the Solomon Islands and Kenya, mangroves have been regarded as sacred spaces where special rites take place, temples are erected, and trees worshipped (Kathiresan & Bingham 2001). In Latin America, the importance of these habitats to indigenous cultures is evident from zooarchaeological data, which supports cultural anthropological inferences about prehistoric subsistence and ecology (Stahl 2003; Wake et al. 2013). Examples of regional mangrove resource exploitation by pre-Spanish peoples are numerous. When present, mangroves were intensively utilized for timber, charcoal, tannins, shell collection, and fishing (Prahl 1989; Lacerda et al. 1993). Species that frequent mangroves often represent a large biomass accessible from the land, and can be readily harvested by fairly simple techniques (e.g. by hand, and with weirs and traps) (Cooke & Jiménez 2004). This generated a steady source of animal protein and enhanced human population growth, sedentism and ultimately societal complexity. Dating from the early Formative Period (1800 BCE–200 CE), the coastal culture of Valdivia (Ecuador) holds evidence not only of broad-based fish and marine molluscs capture from mangrove habitats and shallow intertidal waters, but of hunting for mangrove associated birds like grebes, ibises, ducks and coots (Stahl 2003). In central Panamá, aquatic resources and particularly fresh and preserved fish contributed substantially to the diet of pre-Columbian societies (Cooke et al. 2008).

In the early indigenous communities' transition from nomadic to sedentary living, resource rich habitats such as estuarine mangrove are thought to have played a fundamental role (Prahl 1989; Lacerda et al. 1993; Raymond 2008). The earliest and most complete records of permanent settlements have been discovered in the small coastal valleys of south-western Ecuador (Raymond 2008). Using carbon dating of fossilized *Cucurbita* fruit (squash and gourd), Piperno and Stothert (2003) were able to identify an early pattern of agriculture in coastal Ecuador that dates back to the Early Holocene (10-12,000 calendar years ago). Relevant early agricultural sites were located at the interface between marine coastal, fluvial and mainland habitats. Therefore, fertility and diversity of coastal low lands adjacent to mangroves might have provided a suitable place to hunter-gatherer communities in coastal Ecuador while they started cultivating edible plant varieties, on their way to establishing fairly sedentary occupations (Piperno & Stothert 2003). Evidence of pre-Columbian use of mangrove-estuarine resources has been found throughout the ETP: Reitz and Masucci (2004) found that the main invertebrates in deposits of the Ecuadorian coastal settlements, Valdivia and Machalilla, are mangrove-associated molluscs such as the horn shell *Cerithidea pulchra* and the blood cockle *Anadara tuberculosa.* Between 700 BCE and 500 CE, the culture called “Tumaco/Tolita” situated between Buenaventura (Colombia) and Esmeraldas (Ecuador), harvested mangrove products, such as molluscs, fish, crabs, birds, and mammals (Villegas et al. 1994; Zuluaga & Romero 2007).

Pre-Columbian societies took advantage of the biologically diverse ichthyofauna of the ETP by exploiting a wide range of species, mostly marine inshore and euryhaline freshwater fish found in mangrove channels and low salinity shallow waters using tidal traps, weirs, and perhaps hook and line (Cooke & Jiménez 2004). The importance of fish in the diet is evident in archaeological sites of the Coclé culture from the lowlands of central Panamá, where the targeted fish taxa (e.g. catfish, sleepers, snook, toadfish and croakers), point towards a fishing strategy that focused on intertidal mudflats, mangrove forests and tidal rivers (Cooke & Ranere 1999). However, not only coastal settlements benefited from the abundant fish resources: In one of the most populated zones in pre-Columbian Panamá, the littoral and adjacent wooded savannahs of Parita Bay (a mangrove fringed estuarine system), marine fish bones have been recovered in sites located 13 – 60 km from the coast (Cooke & Jiménez 2004). Cooke and Ranere (1999) found that 70% of the fish consumed between 1500 - 1800 years ago in a site 13 km away from the Parita Bay coast, were of marine origin. These included many that frequent mangroves, but also others that eschew this habitat and favour clearer water currents at the seaward edge of the turbid estuarine mixing zone. According to ethnoarchaeological data, it is suggested that fish was preserved to be exchanged with inland communities by salting and sun-drying (Zohar & Cooke 1997; Carvajal-Contreras et al. 2008). The facility of sun-drying salted fish for local and regional consumption, and its subsequent distribution inland, is apparent at the Vampiros rock-shelters on an ephemeral ancient strand line in Parita Bay (Carvajal-Contreras et al. 2008).

The above summary underlines the potential of archaeozoological research in Latin America to provide detailed data about how pre-Columbian societies interacted with the coastal habitats through time, in tandem with substantial diachronic geomorphological changes that affect mangrove extent, accessibility and distribution. Much information, however, remains to be uncovered in this field. This type of research is challenging given that: 1) the conservation of archaeological evidence is impaired by the instability and ephemerality of relevant coastal landforms in time and space (Clary et al. 1984; Cooke & Ranere 1999); 2) the difficulty of accurate identification of species, especially in speciose families and genera with heterogeneous life histories, and 3) the scarcity of qualified researchers in fish biology and archaeoichthyology in most countries (Cooke & Martin 2010).

2.2.3. Conquistadores and the colony

The Spanish were the first to provide written accounts and descriptions of the mangroves of the American continent during their expeditions. Some coastal pre-Columbian societies used mangrove wood, apparently preferentially. In the early stages of the colonial period, however, the Spanish intensified the exploitation of mangrove wood by utilising it heavily for construction, especially shipbuilding, because of its water resistant qualities, hardness, length, and girth (De Ulloa & Juan 1826). Timber harvesting played an important role in construction and leatherwork, while mangrove charcoal was used in sugar production. For these reasons, mangrove wood became part of the tax or 'tribute' that the indigenous communities had to pay the Spanish king (Figure 2.2) (De Ulloa & Juan 1826; Prahl 1989).

During the 17th century, the Spanish were eager to broaden their naval domain and promoted the construction of shipyards in strategic cities of Ecuador, Costa Rica and Panamá (Guayaquil, Nicoya, Ciudad de Panamá) (Jordán Reyes 2006). These demanded large quantities of wood, such as Tabebuia *sensu lato,* mangrove, and laurel (*Cordia spp.).* The Spanish monarchy claimed that the Guayaquil shipyard was the most important of the Pacific coast of the Americas because of the quality of its ships (De Ulloa & Juan 1826). Between the 16th and mid-18th centuries, the demand for wood for the shipyard and for the construction of churches and buildings in Lima was so high, that mangrove poles exported from the Pacific of Colombia reached six thousand poles per year (Prahl 1989). Mangrove wood from Ecuador (Esmeraldas and Guayaquil regions) was also exported to Perú to build coastal cities such as Lima, because of the lack of forests in this region (Patiño 1990). Jorge Juan and Antonio de Ulloa mention the exploitation of mangrove wood in Ecuador in their book '*Secret news of America*', published in 1747:

"*In these works* [building and repairing] *they employ great quantities of mangroves taken from Guayaquil annually by the King…the loss of mangroves and workforce* [on the repair of walls]*, rises to very considerable quantities*" (De Ulloa & Juan 1826). Translation by J. Lopez-Angarita

Figure 2.2. "Indigenous exploitation by the Spanish conquerors". A mural by Diego Rivera (1929-1945) showing an indigenous workforce involved in wood extraction (top right quarter). Because of its high quality, large amounts of mangrove wood were utilized as building material. Mexico City - Palacio Nacional. Source: http://tinyurl.com/qdhx3jr

The demand for mangrove wood was so high that the Spanish monarchy were forced to issue regulations for its exploitation, such as licences and permits required for cutting certain species, or in certain areas (Jordán Reyes 2006). After gaining independence from the Spanish crown in the early 19th century, the young Republics were left without an understanding of the importance of managing their forestry resources, while facing political volatility and instability (van Bottenburg 1952). Mangrove wood exploitation became a very important industry on the Pacific coast of the continent, with its main hubs in Buenaventura (Colombia) and Guayaquil (Ecuador). Uncontrolled logging activity continued for many years without any replantation of trees (Cifuentes 2002).

Exploitation and commercialization reached industrial levels after 1948, when two businesses located in Buenaventura monopolized tannin production for the next 30 years. By the 1960's they were producing approximately 3000 tons of mangrove wood per month, mostly red mangrove, *Rhizophora mangle* (Leal 2000). By the 1970s mangrove wood exploitation in Colombia for tannin extraction and construction, had reached its peak, and subsequently collapsed, for two reasons: 1) Prices in Colombia were undercut by the international tannin market, and 2) deforestation levels increased tannin manufacturing effort (Cifuentes 2002). Large trunks were diverted to making power line poles and railway sleepers (Prahl 1989).

For more than 400 years, colonial and republican use of mangrove wood was governed only by profit maximisation, causing widespread deforestation. It wasn't until the 1990s that mangrove forests began to be considered as ecosystems, and managed as such (Lacerda et al. 1993). In Costa Rica, mangroves started to be impacted by coastal development in the early 1940s when the country's population underwent a rapid rise and began to convert large areas of mangrove stands to agriculture, aquaculture, and wood extraction (FAO 2007b). By 1982, Ecuador had the world's largest area dedicated to shrimp production, and in only 30 years following the construction of the first shrimp ponds in 1969, 57% of Ecuador's mangroves had been cleared for shrimp farming (Ocampo-Thomason 2006). This was the result of the high international demand for shrimp and the economic incentives provided by the government (Martinez-Alier 2001).

Panamá was one of the first countries in Latin America to establish a commercial shrimp farming industry (Bolanos 2012) and after shrimp aquaculture began in 1974, production grew rapidly with 8100 ha under production in 1998 (Suman 2002). However, despite mangrove-lined channels being often eradicated, the most impacted habitat was that of high tidal flats or salt flats (*albina* in local Spanish) publicly owned and adjacent to mangroves in the central Pacific provinces of the country (Suman

2002). Into the 1990s mangroves were still important assets in Costa Rica and Panamá for tannins and charcoal production (Lacerda et al. 1993).

Nowadays, mangrove clearance or other anthropogenic modification is primarily related to aquaculture, agriculture and urban land uses. Timber extraction still causes degradation of remaining forests, although newer threats include solid waste disposal, pollution, rising sea level and overfishing (Hogarth 2007). Where mangroves are close to urban areas, their conversion to a constructed environment (e.g. housing, ports, and industries) is widespread (Benfield et al. 2005). Other areas are transformed into arable and grazing land (Giri et al. 2008). However, the most important driver of mangrove loss is aquaculture. Hamilton (2013) found that commercial aquaculture accounts for 28% of mangrove lost since 1970, in 8 of the countries with highest mangrove area globally. Vietnam lost 75% of its mangroves between 1968 and 2003, and aquaculture accounts for 40% of those losses (Binh et al. 2005). Similarly, aquaculture is responsible for 63% of mangroves lost in Indonesia between 1975 and 2005 (Giri et al. 2008). In Ecuador, mangrove clearance between 1970 and 2006 has been attributed almost entirely to shrimp farming (Hamilton & Stankwitz 2012; Hamilton 2013). Particularly, extensive shrimp aquaculture~~, which~~ is often related to unsustainable practices involving the use of 1) fishmeal production from wild-caught fish, driving overfishing and associated bycatch to feed shrimp, and 2) fungicides, pesticides and antibiotics, which pollute ground water and damage soil leading to pond abandonment and prevention of recolonization by mangroves (Páez-Osuna et al. 1998; 2003).

2.3. Change in mangrove extent

Obtaining reliable estimates of long-term changes in the areal extent of mangroves is compromised by the lack of data and large variance of area estimates. Significant differences between each country's estimates are apparent. These discrepancies in part reflect the difficulties in arriving at accurate estimates through mapping (Heumann 2011; Kuenzer et al. 2011; Friess & Webb 2014; Hamilton & Casey 2016). By 1999 it was suggested that certain Latin American countries had lost up to 40% of their total mangrove area (Lacerda & Schaeffer-Novelli 1999). However, precise estimates of mangrove deforestation are still lacking for the region due to the inherent difficulties in establishing a baseline. Only since the late 1990s has satellite spatial imagery been used for systematically mapping natural resources at a global scale (the *Enhanced Thematic Mapper Plus* system of Landsat was launched in 1999) and recent advances in remote sensing technology have facilitated the availability of higher resolution estimates (Spalding et al. 2010; Hamilton & Casey 2016). Even so, many present-day mangrove formations are narrow or patchy, and blend subtly with other habitats, making them hard to detect using satellite data imagery, and estimates prone to error (Manson et al. 2001).

To determine the change of mangrove coverage in the ETP we collected published estimates of total mangrove forest area for Panamá, Costa Rica, Colombia and Ecuador. We compiled all estimates available in the literature, from the earliest to the latest, regardless of the detection method (Prahl 1989; Lacerda et al. 1993; Guevara-Mancera et al. 1998; FAO 2007b; 2007c; Giri et al. 2011) (Appendix 2.1). In some cases more than one annual estimate was reported (by different sources). Here we averaged the estimates and used the standard error as an indicator of precision for the different estimates. For our calculations, we used the dates of the dataset rather than the date of publication of the estimate, unless the dataset date was not specified. Including all the available figures of mangrove cover will likely increase the error margin of the estimation of trends, however, given the scarcity of historical mangrove area information we decided the possible bias was justifiable.

We estimated the change in mangrove area by country and the rate of mangrove loss by calculating the difference between the earliest and latest available estimates of mangrove area. We then divided the change in area by the number of years between the latest and earliest estimate (Table 2.1).

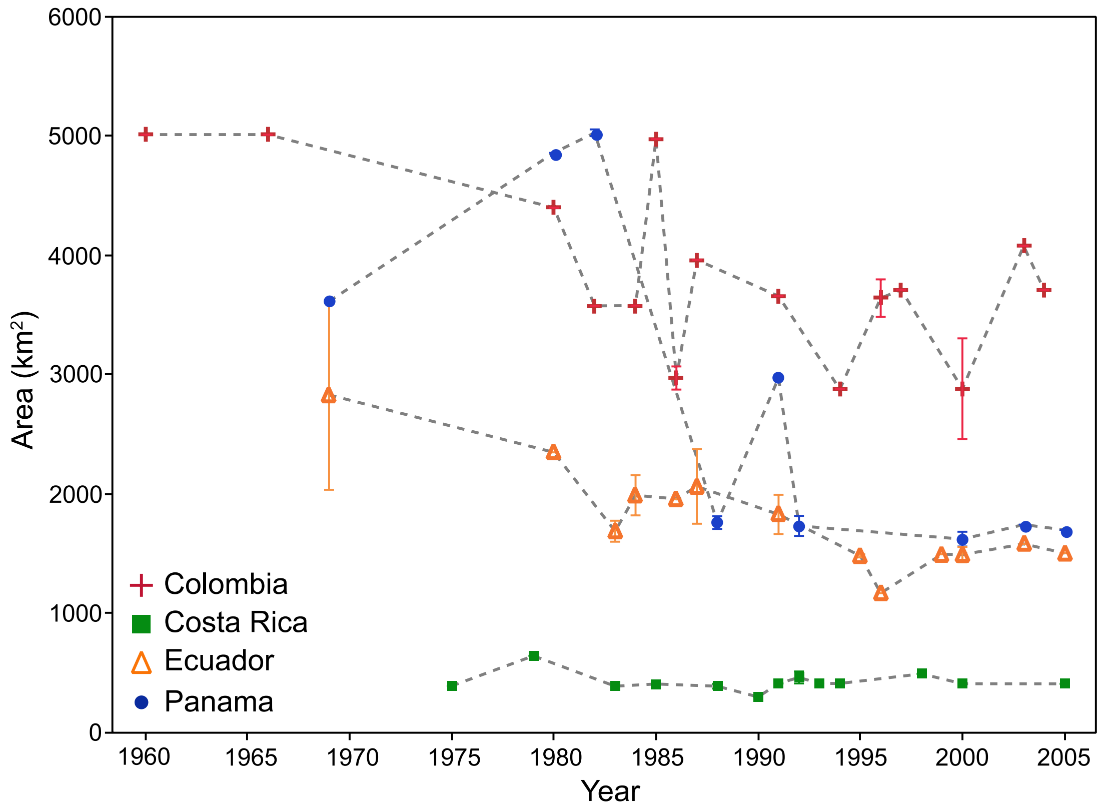

Figure 2.3. Change of mangrove area from 1960 to 2005 in the Eastern Tropical Pacific. The graph was constructed using all available mangrove area estimates in the literature. Error bars are calculated using one standard error from the mean for years with more than one independent estimate.

We found high variability among published estimates of mangrove area with notable oscillations in time, and a high standard error for area measurements estimated in different sources for the same year (Figure 2.3). From this we assume that the irregular results were due to the different techniques being used to conduct year-by-year area measurements since the precision and accuracy of remote sensors can vary significantly depending on methods and mapping objectives (Mumby et al. 1999). As our results represent all existing estimates of the region, they should be interpreted with this in mind. Some values are likely to have underestimated or overestimated the real coverage, however, despite the variability of the data, there is a clear trend of mangrove area decline. From the rate-of-loss data (Table 2.1), it seems that Panamá has experienced the greatest overall loss of mangrove cover in the region, followed by Colombia and Ecuador. Costa Rica shows the highest proportion of intact mangrove forest, but also has the smallest mangrove area of the four countries.

Table 2.1. The most recent and first available estimates of mangrove area, the percentage lost, and the mean annual rate of loss for the four countries of the Eastern Tropical Pacific.

Country	Mangrove area (ha)		Percentage loss of mangrove area (ha)	Annual rate of loss (ha y^{-1})	Annual percentage loss rate
	Most recent estimate* (Giri, et al 2011)	Earliest estimate* (year)			
Panamá	154,227	486,000 (1980)[1]	68.2	16,588	3.41
Ecuador	137,698	362,727 (1969)[2]	62.0	7,259	2.00
Colombia	213,857	501,300 (1960)[3]	57.3	7,186	1.43
Costa Rica	39,034	64,452 (1979)[4]	39.4	1210	1.88

**The year of the earliest and recent estimate provided correspond with the most reliable estimate in relation to the data, hence some years do not match those of Figure 2.3.*

[1,4] FAO. 2007. Mangroves of North and Central America 1980-2005: Country Reports. Forestry Department. FAO. Forestry Department. FAO, Rome.

[2] Ocampo-Thomason, P. 2006. Mangroves, People and Cockles: Impacts of the Shrimp-Farming Industry on Mangrove Communities in Esmeraldas Province, Ecuador. Pages 140–153 in C. T. Hoanh, T. P. Tuong, J. W. Gowing, and B. Hardy, editors. *Environment and Livelihoods in Tropical Coastal Zones.* CAB International.

[3] Villalba, J. C. 2005. Los Manglares en el Mundo y en Colombia. Sociedad Geográfica de Colombia Academia de Ciencias Geográficas:1–22.

Neighbouring Panamá and Costa Rica showed the greatest difference in the magnitude of mangrove area loss. Costa Rica has a strong tourism sector and thanks to the high proportion of protected areas (World Economic Forum 2013) and national laws in favour of ecosystem conservation, has maintained a successful ecotourism industry (Krüger 2005). Therefore, total area of mangrove lost in Costa Rica is low compared to the other countries in the region, as legislation has proven effective in general (Jiménez 2004; Spracklen et al. 2015). On the other hand, Panamá has a growing infrastructure sector, with urban areas spreading rapidly into natural areas such as wetlands, despite protection policies in place (Kaufmann 2012) **(Box 1)**. The greatest proportion of mangrove destruction in Panamá has occurred around Panamá City where coastal wetlands have been heavily disturbed in the last two decades as the city grows and land is reclaimed from the sea (Kaufmann 2012). Beach areas that are near small patches of mangroves, i.e. Punta Chame to San Carlos, have also suffered mangrove loss for tourism development. Many Panamánian environmental organizations, government and non-profits, are vociferous about the dangers of mangrove destruction, but political corruption too often inhibits protection and conservation action (Mate 2005).

Historical losses of mangrove cover will never be fully appreciated (Alongi 2002), but even the recent reported losses of coverage in the ETP highlight the urgent need to strengthen information systems and obtain reliable figures on which to base future estimates and conservation measures. The best estimates rely on a combination of remote sensing images, aerial photos, forest surveys, and ground-truthed maps, yet for remote and inaccessible areas achieving a high accuracy is still very challenging (Manson et al. 2001; Friess & Webb 2014).

Mangrove losses can be very costly, especially for developing countries, and are the result of our failure to link ecological processes with their societal and economic benefits. The need to protect mangroves with effective conservation measures is best advocated by the negative impacts recorded after human-caused perturbations to the habitat (Alongi 2002). As valuation of mangrove services has proven to be a useful tool for proposing a more sustainable use of wetlands, we have been able to understand more clearly how costly are the ecological and social implications of mangrove loss (Naylor et al. 2000; Valiela et al. 2001; Walters et al. 2008), and have discovered some of the prominent economic benefits that may represent hope for the future conservation of mangroves (Rönnbäck 1999; Aburto-Oropeza et al. 2008).

2.4. The state of mangrove protection in the ETP

In Latin America and the Caribbean, Guarderas et al. (2008) found that, despite the increase through time in the number and area of Protected Areas only 1.5% of the coastal and shelf waters of the region are under some type of conservation protection (percentage of exclusive economic zone protected: Costa Rica = 1%, Panamá = 13.5%, Colombia = 9%, Ecuador = 13%) (Appendix 2.2). Since lack of protection can be extremely costly in terms of loss of ecosystem services (Tallis & Kareiva 2005), there is a great need to include highly valuable coastal wetland habitats such as mangroves within protected schemes. Polidoro et al. (2010) found that the highest proportion of threatened mangrove species in the world occur in Costa Rica, Colombia and Panamá, with 25 to 40% of mangrove species classified as threatened under the IUCN Red List Categories of Critically Endangered, Endangered and Vulnerable (**Box 2**).

Despite the high mangrove biomass found in the ETP (Hutchison et al. 2014), mangrove tree species diversity within the region is low relative to other regions (**Box 2**), making it particularly vulnerable to species loss, and consequently, the effects on human livelihoods and ecosystem services are expected to be greater than in other regions with higher diversity (as systems with higher regional species richness are argued to be more stable) (Worm et al. 2006). Moreover, the ETP suffers from significant gaps in protected area coverage compared to other regions such as the Caribbean, as well as little connectivity between existing protected areas (Guarderas et al. 2008). These are factors that highlight the need to focus on designation of a protected area network in the region, rather than single reserves established in isolation (Guarderas et al. 2008).

To explore the characteristics and management strategies of the protected area system in the ETP, we consulted the literature and available information in government websites to create a list of those protected areas that include mangroves and are formally recognized by each national authority. We conducted a review of the management plans of the protected areas identified and, given the multiple management categories found, we classified the protected areas following Guarderas et al. (2008) into: no-take protected areas, limited-take protected areas, and mixed-use protected areas (a no-take area within a limited take area). To determine how well mangrove habitats are represented in these protected areas, we used data from Giri et al. (2011), representing the latest spatial information of global mangrove coverage available from the World Conservation Monitoring Centre (WCMC). Giri et al. (2011) used 1000 Landsat images, ground-truthed data, and published literature to estimate the global distribution of mangroves. Mangrove area calculations were performed inside and outside protected areas using polygons in ArcGIS 10.2 (ESRI). We estimated the proportion of protected mangroves using maps of the protected areas obtained from *Protected Planet* (www.protectedplanet.net) the online interface of the World Database on Protected Areas, combined with national databases from government and non-governmental organisation databases.

According to our sources, there are fifty-one protected areas in the ETP that include mangroves and these cover a wide range of management categories and schemes (Appendix 2.3). Twenty-two occur in Costa Rica, 15 in Panamá, 10 in Ecuador and 4 in Colombia (Figure 2.4). Protection schemes range from national parks to wildlife refuges and ecological reserves managed by local communities. There are marked differences in the management approaches of each of the countries. Costa Rica and Panamá have a higher proportion of no-take protected areas than Ecuador and Colombia. In Costa Rica coastal wetlands are all no-take areas (RAMSAR 2001) (Table 2.2), so any protected areas that allow use of non-wetland natural resources and have mangroves within their limits are classified as mixed-use protected areas. Mangrove representativeness results show that at a regional level 47% of the total mangroves of the region are included within protected areas. Costa Rica and Ecuador lead the region in mangrove protection, with 58.7% and 53% respectively, whereas Colombia, at 28%, has the lowest proportion of protected mangroves in the region.

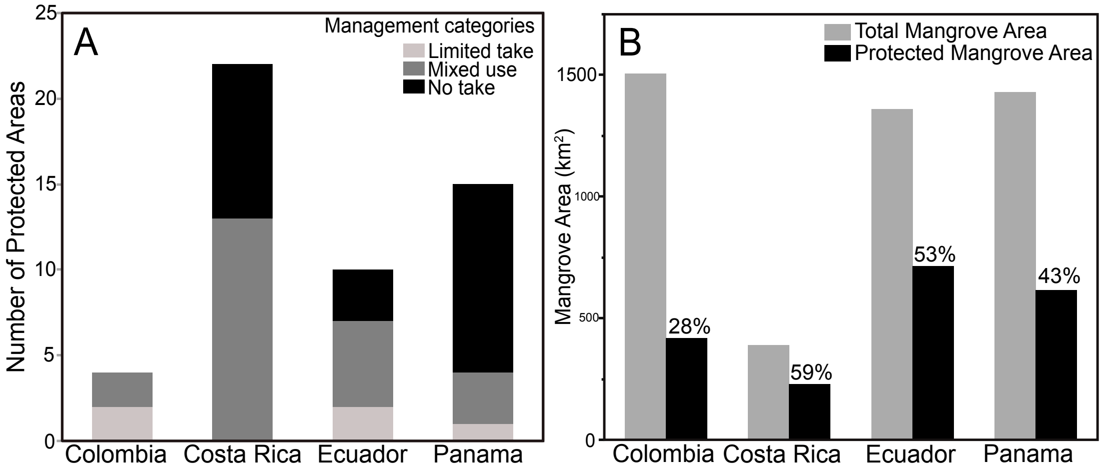

Figure 2.4. (A) A stacked histogram showing the number of protected areas located in the Pacific coast with mangroves per country by management categories of no take, limited take and mixed use. (B) Percentage of Pacific mangrove forest area included in protected areas by country.

Among the key protected areas in the region, which conserve a high proportion of mangrove area are the Terraba-Sierpe RAMSAR wetland in Costa Rica (Figure 2.5) with an extension of 30,000 ha, Sanquianga National Park in Colombia with 80,000 ha, and the Ecological Reserve Manglares Cayapas Mataje in Ecuador with 51,300 ha. As a general trend across the ETP, there is poor documentation on the governance of protected areas and many still lack management plans. Additionally, the administrative structure of protected areas were originally designed for terrestrial areas, but recently governments have increased the representation of marine areas and habitats in protected areas (Alvarado et al. 2012). Official agreements for marine conservation facilitate the implementation of participatory management schemes for protected areas (García 2010), with the aim of generating economic alternatives for communities, such as ecotourism (TNC 2011). Evidence of participatory management processes and schemes during the creation and administration of the protected areas are limited, even though governments officially recognize their importance (Fundación Futuro Latinoamericano 2011; Vieira et al. 2016).

2.5. Environmental policy related to mangroves

We reviewed all existing environmental policies related to mangroves at international and national levels, by searching the literature, soliciting legal documents from government offices, and consulting their websites (Table 2.2). We found that as recently as 30 years ago, wetlands and especially mangroves were considered unproductive land by local governments. The ignorance of the value of mangroves in terms of provision of ecosystem services, or merely the attraction of short-term financial gains to be had from developing mangrove land, enhanced their destruction and clearance. Frequently governments and multinationals like the World Bank and the International Monetary Fund encouraged this attitude (Yañez-Arancibia & Lara Domínguez 1999; Ocampo-Thomason 2006; FAO 2007c; Warne 2011). This situation started to change in the 1990s as a result of increased global awareness of ecosystem services, and soon governments of the ETP joined international movements towards more environmentally oriented policies through international initiatives, such as The Ramsar Convention on Wetlands and the Convention on Biological Diversity (CBD).

Figure 2.5. Mangroves of Costa Rica showing an aerial view of a mangrove fringed river mouth in Golfito (left panel) and the mangrove forests of Terraba–Sierpe protected area (right panel). Photo © López-Angarita, J.

Regional agreements emerged later such as the Central American Policy for the Conservation and Rational Use of Wetlands (2002), a common working agenda to strengthen the conservation and sustainable use of wetlands through regional cooperation and action. In South America, the Permanent Commission of the South Pacific created an action plan for the protection of coastal areas of the region (1981). Decades later, in 2004, the UNESCO declared the Marine Conservation Corridor of the Eastern Tropical Pacific (http://whc.unesco.org/en/seascape/) aiming to support the sustainable use of marine natural resources in the ETP, through the establishment of joint regional policy strategies that are supported by the community at large, international cooperation mechanisms and non-governmental sectors (www.cmarpacifico.org). The countries in the ETP have different approaches to mangrove protection policies ranging from full protection to managing them as crucial components for human livelihoods (summarised in Table 2.2).

Table 2.2. A summary of the international conventions, regional agreements, and national laws, policies and regulations related to mangrove habitats for each of the countries of the Eastern Tropical Pacific.

Policy description	Costa Rica	Panamá	Colombia	Ecuador
International Level Ramsar Convention on Wetlands	Approved in 1991	Approved in 1989	Approved in 1997	Approved in 1991
Convention of Biological Diversity	Ratified in 1992	Ratified in 1992	Ratified in 1992	Ratified in 1992
Regional Level Regional agreements involving mangroves	Central American agreement for the protection of the environment (1989)	Central American agreement for the protection of the environment (1989)	Permanent Commission of the South Pacific includes an action plan for the protection of the coastal areas of the region (1979)	Permanent Commission of the South Pacific includes an action plan for the protection of the coastal areas of the region (1952)
	Marine Corridor of the Eastern Tropical Pacific (2004)	Marine Corridor of the Eastern Tropical Pacific (2004)	Marine Corridor of the Eastern Tropical Pacific (2004)	Marine Corridor of the Eastern Tropical Pacific (2004)
	Central American policy for the Conservation and Rational Use of Wetlands (2002)	Central American policy for the Conservation and Rational Use of Wetlands (2002)	No regional agreement specifically on wetlands	No regional agreement specifically on wetlands
National Level Definition of wetland, land tenure laws	1992. Wetlands are public assets with multiple uses.	Mangroves are public assets	Mangroves are public assets	Mangroves are public assets
Policies and laws relating to mangroves. Exploitation and usage regulations.	- 1940. Wastelands Law, mangrove wood extraction needs specific government approval. - 1996. Forestry Law limits the exploitation and	- 1998. General law for the Environment. Mangroves are given conservation priority as an ecosystem with high biodiversity and productivity.	- 1982. Commercialization of mangrove wood poles prohibited. - 1995. Logging of certain mangrove species restricted or prohibited.	- 1994. Construction of new shrimp farms prohibited. - 2004. Exploitation and logging of mangrove prohibited, for

	logging of mangroves - 1998. Biodiversity Law makes all wetlands protected areas, dedicated to the conservation and protection of biodiversity, soil, and water resources. All exploitation prohibited only research and recreation permitted.	- 2006. The logging, use, commercialization, and modification of any mangrove are prohibited. Only allowed if authority declares the activity as sustainable. - 2006. Highly valuable habitats such as mangroves have conservation priority. - 2008. All wetland areas and particularly mangroves are special zones of management.	- 1996. Any activity that exploits mangrove or associated resources is required to have a special licence. - 1998-2002. National Environmental Policy puts all coastal ecosystems under integrated marine and coastal management. - 1996. Certificates for Forestry Incentives. - 2000. National Policy of the Ocean and Coastal Spaces	all but ancestral communities. - 2004. Forestry Law, license needed to exploit mangroves. - 2007. 1 nautical mile from the coastline declared as zone reserved for species reproduction (with specific regulation and uses). - 2008. Shrimp farms obliged to restore 10-30% of the areas illegally occupied. - 2008. Conservation and management of fragile and threatened ecosystems such as mangroves regulated by the state.
National Environmental authority with jurisdiction over mangroves	1978. Ministry of the Environment and Energy. National System of Conservation Areas (SINAC)	1998. National Authority for the Environment (ANAM) 1994. National System of	1993. Ministry of the Environment, National Natural Parks of Colombia.	Ministry of the Environment of Ecuador. 2008. National System of

		Protected Areas (SINAP) 2006. Panamá Authority for Aquatic Resources	2011. National Authority of Fishery Resources.	Protected Natural Areas.
National research and conservation initiatives	- 1993. National Strategy for Wetlands. - 1999. National Wetland Programme. - 2001. National Policy for Wetlands	2004. Inventory of continental water bodies with emphasis in fishing and aquaculture. 2010. Inventory of continental and coastal wetlands of the Republic of Panamá	- 1990. Mangroves of Colombia. - 1990. Conservation, Management and Use of the Mangroves of Colombia. - 2000. National Programme for Mangroves	1984. Temporal and spatial study of mangroves, shrimp ponds and salinas. This study was updated in 1987, 1991, 1995, 1999, 2004, and 2007. 1985. Inventory of mangroves in the Continental Ecuador.
Management strategies	1998. All wetlands are protected areas. Most Ramsar sites have management plans.	2003. The Aquatic Resources Authority issues special permits to guarantee the sustainable exploitation of mangroves.	1991. Afro-descendant and indigenous communities have authority to manage their traditionally occupied lands.	2000. Agreements of sustainable use of the mangroves with ancestral users.

2.6. Prospects for the mangroves of the ETP

For a long time mangroves were considered tantamount to waste lands in Latin America because governments failed to understand their ecological significance. Nor did they understand their great importance for local subsistence economies and, ironically, lucrative export resources such as shrimp. A shift in attitudes began to appear in the 1990s in the face of increasing numbers of scientific investigations that demonstrated the utility of mangroves for human well being (Lacerda et al. 1993).

Concomitantly, Latin American environmental policy underwent a transformation and approved many international agreements (e.g. CBD, RAMSAR, CITES), which led to the modification of political constitutions. New environmental legislation was passed, and several conservation and research initiatives started with the assistance of international agencies and NGOs (Ministerio de Medio Ambiente 2002; CREHO 2010; SINAC 2010; Columba 2013). Currently the appearance of much new legislation and of the proliferation of protected areas, bear witness to the political will of Latin American countries to conserve their mangroves. However, coastal deforestation persists in the region despite of the emplacement of protection mechanisms such as protected areas and national environmental policies (**Box 1**). Drivers of mangrove clearance show substantial regional and national variations (Hamilton 2013). In South East Asia conversion to aquaculture has historically been recognized as the main cause of mangrove loss (Valiela et al. 2001) but agriculture has been shown to play an important role as well (Giri et al. 2008), with rice and palm oil agriculture expansion increasing in recent years (Richards & Friess 2015). In the ETP (excluding Ecuador), where aquaculture is not as prevalent as other regions, more research is needed to determine the land use changes that drive mangrove forest clearance, but evidence points at agricultural expansion as an important driver in the region (Chapters 3 and 5). Other noteworthy problems persist, such as the lack of resources for implementing new policies, weak institutional platforms, and the need for qualified personnel (López Angarita et al. 2014). In addition enforcement is often not effective (García 2010; TNC 2011; Alvarado et al. 2012). Even though the new legislation is laudable and often effective, it is imprudent to assume that illegal activities have ceased in mangroves. They have not. There are still illegal activities threatening mangroves in all nations, highlighting the need for institutional capacity to support legislation (FAO 2007c).

Natural resource management of interface habitats such as mangroves can be highly challenging given the multiple sectors with jurisdiction over them (e.g. fisheries, forestry, agriculture, urban development, transport), all with differing agendas and positions on their use (Castellanos-Galindo et al. 2017) (**Box 1**). This translates into frequent conflicts of interests between sectors, often disregarding mangrove conservation, a risky situation given the strong cohesion and association of mangroves and local livelihoods. In coastal communities with strong ecological and social linkages conservation has a higher economic value than any form of destructive exploitation, such as logging or aquaculture. These linkages are bound to be stronger in highly connected regions such as the ETP, where non-destructive mangrove exploitation represents the main livelihood activity for many local communities. The ETP region needs to protect mangroves effectively, with tangible actions and accurate figures, by taking advantage of the existent regional agreements and commitments, homogenizing political barriers, and framing sustainable development objectives at a regional level.

BOX 1. Panamá City's disputed treasure

The earliest mentions of Panamá City are found in Spanish chronicles from the 16th century, which describe it as "sick" given its location on "a lagoon of foul smell" with rivers "filled with crocodiles", where the Spanish conquerors used to feed on "the great quantity of clams" (de Cieza de León 2005). Authors describe the wetlands of Panamá Bay with a negative connotation common until the last decades of the 20th century (D'Croz & Kwiecinski 1980). Thanks to awareness regarding the importance of the ecosystem services that mangroves provide to the capital, Panamá Bay was designated a RAMSAR wetland site in 2003 and a protected area in 2009 (CREHO 2010).

In April 2012 the supreme Panamánian Court of Justice approved a "provisional suspension" of the resolution that created the protected area, reducing the rates for mangrove logging permits for commercial projects and illegal logging penalties[1]. Later in May 2012 the Ministry of Development approved construction of exclusive residential areas inside the protected area. Several environmental protection organizations and citizens rejected the Court's decision and criticized it heavily, while protests were held and demands made for re-establishment of the Bay's protection status. This pressure was effective and on April 2013, the Court lifted the former suspension of the protected area, emphasizing that the suspension contradicts Panamánian laws for the protection of wetlands (*Entrada No. 123-12*). However, the constant conflict of interests competing for the bay's land, create a continuous legal tug of war for the protected area[2].

The case of Panamá Bay is a clear example of how mangrove destruction can continue, despite existent international agreements and national laws protecting mangroves (Castellanos-Galindo et al. 2017). Examples like this exist all over the ETP, where wetlands are bought for private use (agriculture, rice and oil palm) without public consultation or in complete defiance of their protected area status[3].

Footnotes:

1. *http://www.ancon.org/index.php?option=com_content&view=article&id=452:rechazo-total-a-nefasta-decision-de-la-corte-contra-area-protegida-bahia-de-Panamá&catid=102:notas-actuales-ancon&Itemid=225*

2. *http://www.aida-americas.org/es/refdocs/1843*

3. *http://www.aida-americas.org/es/project/el-humedal-bah%C3%AD-de-panam%C3%A1-mantiene-su-condici%C3%B3n-de-%C3%A1rea-protegida*

<u>BOX 2: Mangrove diversity in the ETP</u>

There are two major floral realms widely recognized in patterns of mangrove distribution: the Indo West Pacific (IWP) and the Atlantic East Pacific (AEP). The IWP comprises 57% of the global mangrove area, is rich in species (62) and extends from East Africa eastwards to the Central Pacific; whereas the AEP that encompasses all of the Americas, West and Central Africa only hosts 12 species in 43% of the global mangrove area (Spalding et al. 2010).

The major differences in floral composition of mangroves in tropical America started to develop after the closure of the Panamá isthmus 3.1 million years ago, which separated the Pacific from the Atlantic. Afterwards very particular climatic processes (dry seasonal climate) that started in the Miocene era gave shape to the actual flora which has two distinct groups of mangroves: species restricted to seasonal dry climates, and species restricted to high precipitation climates (Jimenez 1999). In Panamá and Costa Rica there are marked dry and rainy seasons in certain areas, but precipitation shows a strong spatial pattern depending on topography. In zones with high terrestrial runoff, mangrove communities can be very extensive and diverse. Examples of this are the *Golfo de Nicoya* and the Térraba-Sierpe delta in Costa Rica, and *Golfo de Chiriquí* and *Golfo de San Miguel* in Panamá. The largest areas of mangroves in western South America can be found in the humid coastline of the Colombian Pacific and the north of Ecuador (Esmeraldas region), whereas dryer areas in southern Ecuador have limited mangrove cover (Spalding et al. 2010).

In the ETP, mangrove swamps are generally more abundant on the Pacific coastline as many estuaries, bays and rivers provide a suitable environment for development of extensive coverage, contrasting with the sandy, high-energy shoreline and narrow tidal amplitude of the Caribbean coast (Spalding et al. 2010). A high number of tidal flats and the significant freshwater input from upstream and precipitation allow mangrove trees of the Pacific coast to grow up to 50 m in height, while only small and stunted trees not exceeding 5 m are found in the Caribbean (FAO 2007a).

Mangroves in Latin America and particularly in the ETP have relatively low plant diversity (Table 2.3), with only 11 species (Yañez-Arancibia & Lara Domínguez 1999). However, as an ecosystem they provide great structural complexity creating highly diverse environments that provide multiple ecosystem services. In the ETP the most common genera of mangroves are *Rhizophora* and *Avicennia*. These two genera are widely distributed in the continent with the most widespread species being *R. mangle* and *A. germinans* (Lacerda et al. 1993).

Table 2.3. Mangrove species found in the Eastern Tropical Pacific. The IUCN column provides the species conservation status according to the IUCN Red List of Threatened Species: (VU) Vulnerable, (LC) Least Concern.

Species	**Common name**	**IUCN**	**Costa Rica**	**Panamá**	**Colombia**	**Ecuador**
Acrostichum aureum	Helecho de playa	LC	X	X	X	X
Avicennia bicolor	Mangle salado, negro, o prieto	VU	X	X	X	
Avicennia germinans	Mangle negro o iguanero	LC	X	X	X	X
Conocarpus erectus	Mangle Jelí	LC	X	X	X	X
Laguncularia racemosa	Mangle blanco	LC	X	X	X	X
Mora oleifera	Mangle nato	VU		X	X	
Pelliciera rhizophorae	Mangle piñuelo	VU	X	X	X	X
Rhizophora mangle	Mangle rojo	LC	X	X	X	X
Rhizophora racemosa	Mangle rojo	LC	X	X	X	X
Rhizophora x harrisonii	Mangle rojo	-	X	X	X	X
Tabebuia palustris	Mangle marica	VU	X	X	X	
TOTAL			10	11	11	8

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Chapter 3. Modelling the effects of land use on mangroves and small-scale fisheries in the Gulf of Montijo, Panamá

Abstract

The role of mangroves as pivotal providers of ecosystem services, including fisheries productivity, has been widely acknowledged in the last two decades. In Latin America, despite improvements in mangrove protection such as an increased number of established protected areas and RAMSAR sites, mangroves are still threatened by human activities. Here I examine the relationship between land use activities adjacent to mangroves and small-scale fisheries in the Gulf of Montijo, a RAMSAR site in Panamá. Land use around the Gulf was classified by dividing Google Earth imagery of the area into 1km^2 grid squares, which were ground-truthed. Human uses were ranked according to estimated level of impact on mangroves based on 27 interviews with local informants. From this I developed a spatially-referenced cumulative impact model of human activities on mangroves. Simple linear regressions were used to compare mean impact score and distance to the mouth of the Gulf, with fisheries landings from 15 villages. Results showed that the percentage of mangrove-dependent species found in landings was not related to mean impact score of mangroves in the vicinity of each village. The impact model showed that despite the protection status of the Gulf of Montijo, its mangrove forests are affected by localised human activities, particularly agriculture. This work illustrates the inherent difficulties of studying the link between mangroves and fisheries, and highlights the importance of improving existing fisheries data collection protocols in the GoM. Given the importance of fishing for local livelihoods, evaluating the effects of agriculture on mangroves and their associated fauna will be essential for the sustainable management of this RAMSAR site.

3.1. Introduction

In 2009 the total area of mangrove forests in Panamá was estimated at 1,744 km^2, with 3% on the Caribbean coast and 97% on the Pacific (CCDA 2002; CREHO 2010). Precipitation in the country ranges between 1000 to 7000 mm per year, with 60% of runoff draining into the Pacific coast, where tidal amplitude is large (~6m), and prevailing marine sediments are the sand and mud favourable for mangrove development (D'Croz 1993). Here the dominant mangrove species are *Rhizophora mangle* and *R. harrisonii* (D'Croz 1993) and the most complex mangrove systems occur in the Gulfs of San Miguel, Panamá and Chiriquí where canopies can reach 30-40 metres high (FAO 2007).

Mangroves were utilized by pre-Columbian people around 5000 years ago for timber, charcoal, tannins, and fishing, with Ariidae, Carangidae, Clupeidae, Sciaenidae, Batrachoididae particularly targeted (Cooke & Ranere 1999). Today they have been widely cleared for: agriculture, livestock, salt production, aquaculture, urban or industrial development (D'Croz 1993), and direct extraction for tannins and wood (D'Croz 1993; Ibáñez et al. 2005). Panamá is one of the countries that has experienced the highest mangrove deforestation rates in the eastern tropical Pacific, with around half lost in the last 50 years (FAO 2007; ANAMARAP 2013).

Although Panamá ratified the RAMSAR Convention of Wetlands in 1989, it wasn't until a decade later that the country's first mangrove protection appeared in legislation (Cooke & Ranere 1999; López Angarita et al. 2016). Then from 1990 to 2000 the National Authority of the Environment (ANAM) estimated that nearly 6000 ha of Panamá's mangroves were lost during this time. For the Pacific coast, there are 15 Marine Protected Areas (MPA); seven of these MPA were created before 1990, six between 1990 and 2000, and two after the year 2000 (Appendix 2.3). In 2008, all Panamá's mangroves became marine-coastal management areas, where logging, and any use or commercialization of the system is prohibited, except in areas subject to special regimes, such as sustainable resource management zones (ANAMARAP 2013).

The Panamánian government allows commercialization of mangrove products (e.g. charcoal, timber, tannins) in permitted areas via harvesting permits (e.g. the Gulf of Chiriqui). However, the system is poorly enforced and unsustainable harvesting occurs (Trejos et al. 2008). While pond construction for aquaculture is prohibited in all mangrove areas, it is allowed in tide flats or "albinas" with a special permit, and once a shrimp farm is established, 'minor alterations' can then be made to the surrounding forests to install water pumps, channels and sluice gates (Mate 2005). Similarly, mangrove clearance is allowed for tourism development (Law 2. Jan 7, 2006). Such measures clearly contradict the idea that marine resources are national patrimony in Panamá (Law 44, Nov 23, 2006).

Artisanal fishing in the mangroves and estuaries of Panamá targets highly valuable inshore teleost species (e.g. snapper, snook, bass), shrimps and molluscs using low technology gears such as gillnets, beach seines, hand lines, and long lines. Most boats in the artisanal fleet have outboard engines of ~40 HP but lack navigation equipment. Of the 735 fish species present in the Gulfs of Chiriqui and Montijo, 213 occur in mangrove creeks, of which 33 are directly associated with mangroves (Mate 2005). However, the extensive use of small mesh gillnets by artisanal fishers in mangrove creeks and river mouths has caused a major decline in fish stocks in the Gulf of Montijo, with some driven to collapse, such as the snapper fishery (Mate 2005).

Mangroves are of direct livelihood importance to fishing communities living along Panamá's Pacific coast (Trejos et al. 2007a; 2008). Several studies have explored the link between mangroves and fisheries, focusing on the effect of mangrove extent on fisheries production (Saenger & Funge-Smith 2013; Hutchison et al. 2014; Carrasquilla-Henao & Juanes 2016). However, as many of the studies that have found positive relationships use correlation and regression analysis support the link (Turner 1977; Pauly & Ingles 1986; Carrasquilla-Henao & Juanes 2016), they are faced with the criticism of autocorrelation of explanatory factors as an important limitation (Lee 2004; Blaber 2007; Saenger & Funge-Smith 2013). Saenger and Funge-Smith (2013) suggested that when exploring if mangroves underpin fisheries production, the results are likely to be highly variable because of i) the inherent biological variations on the degree species are dependent on mangroves, ii) the impossibility to verify and compare fisheries dependent data, iii) the variation of fish species assemblages at a wide range of scales, and iv) the diversity and synergy of confounding environmental factors.

The present study contributes to the existing literature on mangroves and fisheries by exploring methods to evaluate the impacts of human activities on mangrove forests of the GoM, and testing the hypothesis that areas with less impacted mangroves provide a wider array of mangrove-dependent species (e.g. cockles, shrimp, crabs), which lessens the need to fish offshore, while places where mangroves have suffered badly from degradation, increase the need for offshore fishing.

In this study I aimed to:

1. Identify the most threatening human activities to mangroves using satellite images and ground-truthing.
2. Develop a cumulative model of anthropogenic impacts to classify Panamánian mangrove forests on a scale of impact.
3. Analyse fishery landings data to explore spatial relationships between fishery characteristics and mangrove condition.

3.2. Methods

3.2.1. Study Site

The Gulf of Montijo (GoM)(80,765 ha), located in the south of Veraguas Province on the Pacific coast of Panamá (Figure 3.1), is one of the most important mangrove systems of the country and was designated as a RAMSAR site in 1990 (RAMSAR 1990). Within Panamá, the GoM was declared a protected area in 1994 and holds the management category of "Wetland of International Importance" (Pinto & Yee 2011). This Gulf is an ideal study site because high-resolution spatial data are available, along with statistics for fish landings. Local communities are primarily engaged in fishing, agriculture, and tourism (Pinto & Yee 2011; Ventocilla 2013). The Montijo wetland is complex with many river deltas, beaches, rocky shores, mudflats and mangroves. The wetland is an important habitat for migratory aquatic birds and large mammals such as monkeys and sloths (CREHO 2010), but it is also a very important agricultural region with a large amount of fields dedicated to crop production, mainly rice, in its flood plain (ANAM 2004). Artisanal fisheries of the GoM target finfish, sharks, lobster and shrimp, using mainly gillnets (Vega et al. 2014). In 2012, the government registered 25 artisanal fisheries cooperatives, with 499 associated fishers and 214 boats operating in the GoM (Vega et al. 2014). For the same year, Panamá's fisheries department confirmed there were 776 active fishing licences for the GoM (Vega et al. 2014).

3.2.2. Human activities affecting mangrove forests

3.2.2.1. Classification of human activities in satellite images

Prior to fieldwork, I interpreted and classified human activities and impacts around the GoM from satellite images using Google Earth. I superimposed a 1km^2 grid over Google Earth images (Figure 3.1C) and for each grid cell, a series of human activities were identified *a priori* and classified as either: 1) aquaculture ponds; 2) crops; 3) population centres; 4) deforested patches; 5) cattle grazing field; 6) industrial infrastructure; 7) tourism infrastructure; 8) ports; and 9) land for development. This subjective classification was done for the entire coastline of the GoM, using the highest possible resolution of images to allow for the best interpretation.

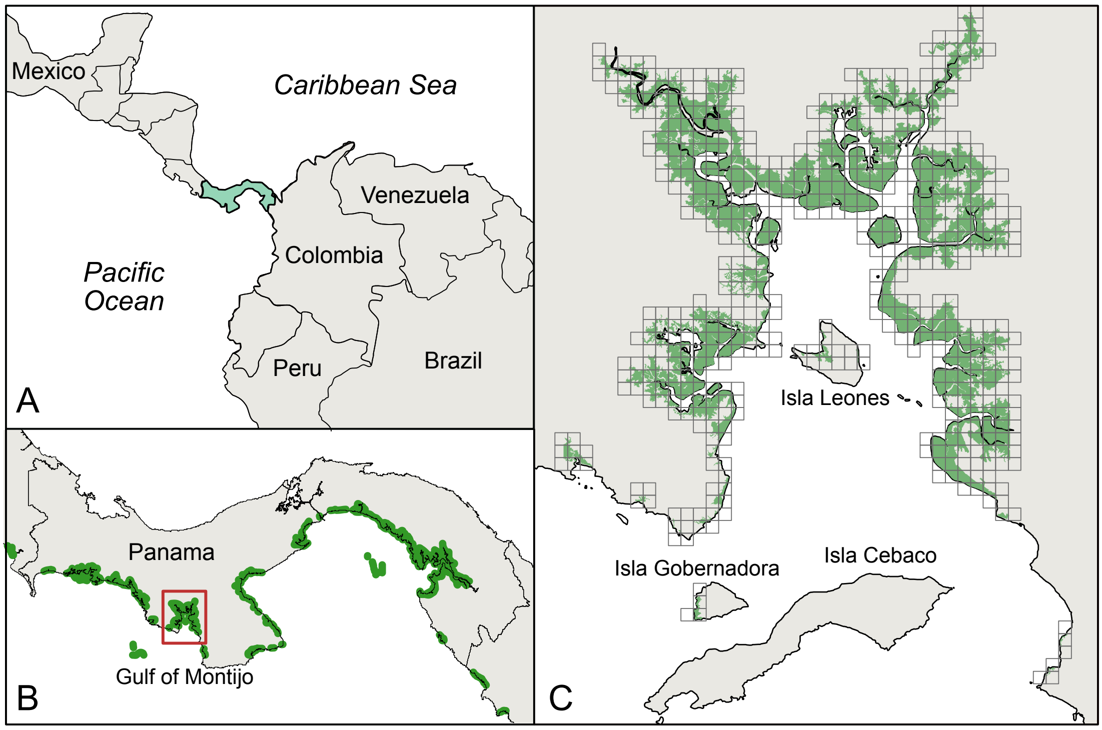

Figure 3.1. (A) The location of Panamá in Central America, and (B) the Gulf of Montijo in Panamá (red square). (C) Illustrates the Gulf of Montijo overlaid with a 1 km grid lines that were used as the sample units to classify impacts along the Gulf's coast. Green colour in B and C represents mangrove forest.

3.2.2.2. Ground-truthing and rapid assessment of mangroves

To calibrate the image-based *a priori* classification of human activities, images were ground-truthed in the field. To select these locations I identified repetitive features in satellite images, as well as features that were blurry or of low resolution. Once in the field, I either went to the identified places in person, or if they occurred on private property or were inaccessible I visually verified and estimated the ground cover and any impact from a nearby vantage point, or contacted the land owner to verify land use. GPS points (Figure 3.2) and *in situ* photographs (Appendix 3.1) were taken as part of the ground-truthing procedure.

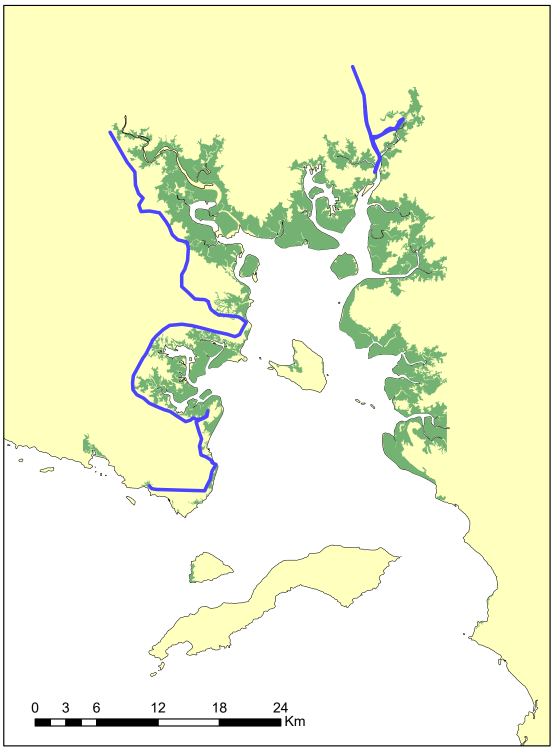

Figure 3.2. Route followed during ground-truthing of anthropogenic impacts in the Gulf of Montijo, Panamá. Pictures were taken and land use information recorded in different points

I visited 12 fishing villages in the area to perform interviews with key informants about ecosystem condition. These people were mostly presidents of fishing cooperatives, experienced fishers (i.e. older than 50 years of age), or community leaders, and twenty-seven interviews were made. Respondents were identified with the help of non-governmental organizations working in the area and by referral of fishermen. Interviews were designed with open questions to gauge the community perception of mangrove condition and their contribution to local fisheries (Appendix 3.2). Questions focused on the respondents' perception of: mangrove health and changes in coverage over time; the identification of human impacts affecting mangroves; the role of mangroves supporting fisheries; and the importance of mangroves for their livelihoods. Interviews were filmed and qualitative analysis was performed using the software Nvivo 10.2. Results were analysed by calculating the proportion of respondents with similar answers per question according to coded topics.

3.2.2.3. Error estimation and calibration

To estimate the error of image interpretation in the GoM I compared *a priori* classification of $1km^2$ grid cells with ground-truthed cells. As any cell could have multiple human activities associated with it, each activity within a cell was scored either 1 (i.e. correctly identified) or 0 (i.e. incorrectly identified). The mean score for each cell was then summed for all ground truthed cells and divided by the sum of all cells at a value of 1. Error was also calculated separately for each human activity.

3.2.3. Mangrove "state" model

Human activities adjacent to mangrove forests pose direct or indirect threats to the ecosystem. To assess human impacts to the GoM's mangroves, I built a model of cumulative impact following Halpern et al. (2008), where human activities that impact mangroves directly or indirectly were overlaid onto a map of mangrove coverage together with detailed layers of land use (Table 3.1). Spatial information for human impacts was obtained from the Marviva Foundation, the Panamá Government Ministry of Environment (ANAM) and the Smithsonian Tropical Research Institute (Table 3.1). Given the information available, I grouped human activities into three major classes: aquaculture, agriculture, and populated centres.

Table 3.1. Source maps and GIS layers used to create a cumulative impact model of mangrove state of the Gulf of Montijo, Panamá.

Layer	***Source***	***Years***
Area of farmed land	Panamá Government Ministry of Environment	2013

Area of aquaculture ponds	Panamá Government Ministry of Environment	2014
Area of populated centres	Marviva Foundation	2000
Mangrove area	Marviva Foundation	2013

The cumulative impact of these activities in mangrove forests was calculated to a resolution of $1km^2$. Grid cell values were checked against the human activities identified during ground-truthing, correcting values where appropriate. Each $1km^2$ was given a score based on the presence (1) or absence (0) of each human activity in the cell. The model calculates a measure of impact by arithmetically adding the scores of human activities within each cell of the grid. I calculated a standard impact score by assigning each activity the same weighting, and a weighted impact score. The un-weighted cumulative impact score (CIS) was calculated for each 1 km^2 cell as follows:

$$CIS_i = Sa_{agri} + Sa_{aqua} + \ Sa_{pop}$$

where *Sa* is the score of each human activity (1=presence and 0=absence), and ***i*** corresponds to each cell of the grid. Subscripts represent the first letters of each activity: agriculture, aquaculture, and populated centres).

For the weighted model, weightings were calculated using a ranking method adapted from Halpern et al. (2007), using scores of mangrove vulnerability to each human impact based on interviews with GoM fishers (Table 3.3). Fishers were asked to identify the most important impact to mangroves of the area. Vulnerability scores were obtained for each impact using the frequency with which that impact was mentioned in responses to determine their relative importance. Frequencies were then normalized. In each 1 km^2 cell, vulnerability scores for each human activity were summed according to their presence or absence. The weighted cumulative impact score (WCIS) was calculated for each 1 km^2 cell as follows:

$$WCIS_i = Sa_{agri}(vs_{agri}) + Sa_{aqua}(vs_{aqua}) + \ Sa_{pop}(vs_{pop})$$

where *Sa* is the score of each human activity (1 = presence and 0 = absence); *vs* is the vulnerability score of the activity; and ***i*** corresponds to each cell of the grid.

For visualization purposes, the model attributes a colour to each 1km^2 grid cell based upon the cumulative impact score, on a green to red scale, representing the state of mangroves according to the influence of adjacent human impacts (Figure 3.3).

3.2.4. Fishery status and mangroves

Fisheries landings information for the GoM was obtained through data sharing agreements with Universidad de Panamá. These data were collected by the Panamá government's fishery resources authority ARAP (*Autoridad de Recursos Pesqueros de Panamá*), and summarizes fishery landings of 15 villages across the GoM between 2008 and 2012. Data recorded include total weight landed per month for a set of 42 commercial species groups identified by common name without detailing species' scientific name, fishing gear or effort (Appendix 3.3, 3.4). In the dataset, some villages have no monthly landings recorded for certain groups. This is because data collection was not systematic, as fishing associations or fisheries leaders of the GoM recorded landings and reported them to regional government offices without any validation (Vega et al. 2014).

The poor quality or lack of fishing effort data is a common limitation found in studies exploring the link between mangroves and fisheries, because without accounting for fishing pressure it is not possible to relate fisheries productivity to changes of mangrove quality/area (Blaber et al. 2000; Saenger & Funge-Smith 2013). Ideally, fisheries independent data should be used, as this will also control for the effects of historical overfishing (Saenger & Funge-Smith 2013). In the GoM, fisheries catch declines have occurred over relatively short timeframes. For example, when the area was designated a RAMSAR site in 1990, the most important fisheries were for snappers (*Lutjanus jordani, L. guttatus, L. chrysurus*), and lobster (*Panulirus gracilis*) (RAMSAR 1990), but these fisheries collapsed around ten years later due to overexploitation (Mate 2005). Therefore, given the uncertainties generated when using fisheries dependent data, my results need to be interpreted with caution.

To compare sampling effort of government data between villages, I created a numerical indicator of sampling effort based upon available data using the proportion of a theoretical optimal sample where all villages are sampled 12 months of the year, and all species present are recorded. Therefore, if all of the species categories were present in the sample, then the ideal sampling effort for each village should have 504 "species-months" per year. To estimate a numerical indicator of sampling effort, I calculated the months sampled per village and divided them by the ideal effort (N = 504). The results were then normalized by dividing by the maximum value in the data.

Despite the limitations of the government dataset, it is still valuable given that it represents the only information available for artisanal fisheries taken throughout the entire GoM. To reduce sampling bias, I used the percentage by number and weight of mangrove and non-mangrove-dependent fishery species per village as response variables for my analyses, calculated across the entire dataset. I defined mangrove dependence following the criteria of Blaber (1989), namely: "*Species for which estuaries or similar habitats are the principal environment for at least one part of their life cycle*".

To allocate mangrove dependency, I identified species using the common names provided on the landings data sheets using confirmed species records existent for the Gulf. For species that were not sufficiently identified or whose life cycles are not fully known, I used information from similar species to classify them as mangrove-dependent or not. In some cases, landings were not identified with common names, but were grouped into categories such as "snappers", "catfishes", "sharks" and "revoltura" (i.e. a mixture of low value weakfish, drums, and croakers) (Appendix 3.3). For these groups, dependency was assigned according to the dominant habitat of the species present in the Gulf belonging to the categories.

To crosscheck the reliability of the government data, I used fishery landings data from a participatory monitoring program in the village of Hicaco between 2012-2014, performed by the Marviva Foundation. Estimated total landings and mean weight of landings per year were compared between the two datasets, as well as species composition of yearly landings by mean weight of fish landed in Hicaco.

The percentage by number and weight of mangrove-dependent and non-mangrove fishery species per village were analysed using linear regressions, against distance of the village to the mouth of the GoM, and the mean impact score of mangroves around villages. The percentage of mangrove species by number was calculated as the proportion of mangrove species found in the total number of species groups present in the landings of each village. The percentage of landings by weight made up of mangrove and non-mangrove species was calculated as the proportion of the total weight of the landings for each village, corresponding to mangrove and non-mangrove species groups.

Distance to the mouth of the GoM was measured as the shortest direct distance between a given village and a line plotted between the east and west sides of the mouth of the GoM (Lat 7.60770, Long -80.97618 and 7.60770, -81.22719). Distances were measured in ArcGIS 10.3.1. Villages situated within the line demarcating the mouth of the GoM were treated as positive values, with villages outside the GoM given negative values. Mean mangrove impact score was calculated per village using scores from the weighted impact model. Since impact score was calculated for 1km^2 grid cells, each village was placed in the centre of a 9km^2 area and the scores of the grid that fell within this area were averaged. The 9km^2 threshold was selected because the score reflects the condition of mangroves in the immediate vicinity of the village. Cells with no mangroves within the designated area were omitted from the mean impact score calculations. Villages without any associated impact score, because of lack of adjacent mangroves, were excluded from the analyses involving mean impact score. An exception was made for Gobernadora Island as the 9 km^2 around the town did not include any mangroves. However, given the location of the island in the mouth of the GoM, I calculated impact score using all the 1km^2 cells present on the island, as mangroves there are most likely to be those influencing the nearby fisheries. The degree to which fishing was considered localised for each community location was based upon informal interviews conducted with fishers at each location. In Puerto Mutis fishers were back from longer trips and spoke of fishing far offshore and well outside the Gulf, so Puerto Mutis was excluded from the analysis.

3.3. Results

3.3.1. Human activities affecting mangrove forests

The main economic activities in the GoM are cattle farming for milk and beef, and rice production predominantly for the national market. Cattle graze in fields with plenty of native vegetation. In addition, on smallholdings, beans, sugarcane and banana are grown for local consumption, but the detail of these is not apparent on satellite images given their small size (*pers. obs.* 2014). From the GoM coast I identified human activities inside 145 1km^2 cells. The easiest human activity to identify was aquaculture, as the ponds have a very distinctive geometrical pattern and the water reflects light when in use, or resembles a bordered but empty "mud field" if not.

After ground truthing, the initial nine impact categories, were condensed to six (Table 3.2), as the categories of crops, cattle grazing fields, deforested patches, and land for development were joined into one single category called "agriculture". Within the cells, I was able to classify human activities with an error rate of 17% (Table 3.2) by assigning a score to each cell depending on whether or not the *a priori* classification was correct. The source of this error only comes from agriculture, as the other land use activities were accurately classified. The error I made was to consider semi-forested fields as plots for land development, or, deforested plots. Following interviews with locals I realised this was actually agricultural land rotated for crop production, cattle grazing, and fallow periods. As this made it impossible to ascertain the real-time use of such land from Google Earth I kept the term "agriculture" as its impact category rather than sub-dividing into more specific land use (i.e. crops, cattle farming).

Table 3.2. Summarized results of testing the accuracy of classifying land use activities from satellite images of the Gulf of Montijo. Error values reflect the percentage of incorrect classifications for each human activity type.

Human Activity	**N***	**Error**
Aquaculture	10	0%
Agriculture	137	17%
Infrastructure	1	0%
Infrastructure (tourism related)	1	0%
Port	1	0%
Town	5	0%

**Sample size is different from the total number of cells ground-truthed because some cells had more than one impact associated with them.*

Twenty-six men and one woman were interviewed from twelve fishing villages in the GoM (Appendix 3.5). All were fishers who had lived their entire lives in the area. Eighty-nine percent were older than 40 and most had been fishing since they were young. 96% of respondents reported that fish productivity had declined in the GoM compared to when they started fishing, and 30% mentioned the collapse of snapper and shrimp fisheries specifically. 46% of respondents felt that overfishing was the main cause of fishery declines with natural fluctuations of fish abundance and climate change also mentioned by 22% and 7%, respectively.

Agriculture was the most important threat to mangroves raised by all respondents (100%), as this had caused their clearance in the past. Likewise, all stated that fumigation and fertilization of agricultural fields, during the dry season, and for rice in particular, caused massive die-offs of fish during the rainy season when rivers wash chemicals into the sea. When asked if they felt mangroves were important to their fishing, 100% of respondents said they were, and most (65%) were aware that this was due to their provision of food and habitat to juvenile fish (i.e. as nursery grounds). Snook (*Centropomus medius*), snappers (Lutjanidae), hammerhead sharks (Sphyrnidae), mullet (Mugilidae), weakfish (Sciaenidae), shrimp (Penaeidae), catfish, cockles (*Anadara tuberculosa* and *A. similis*), and crabs (*Callinectes arcuatus*) were identified as important commercial species that only occur around mangroves. 18% said mangroves help maintain water quality, and 15% said they provide aesthetic value to the landscape. Illustrative comments on these topics included:

> *"Mangroves are everything, as they give life to animals and humans."*

> *"No mangroves, no fish."*

3.3.2 Mangrove "state" model

The results of ground-truthing were used to verify the source maps of land use in 632 1km^2 cells from the grid placed over the GoM. A mangrove state model was generated to incorporate this information. Figure 3.3 shows the resulting overlay from the un-weighted impact score, with four levels of impacts ranging from very high (i.e. 1km^2 cells with all human activities present) to low (i.e. cells with only mangrove present). For the weighted model, all interviewed fishers classified agriculture as the most threatening activity, followed by aquaculture and centres of population (Table 3.3). Impact score in each cell was calculated according to the vulnerability scores (Appendix 3.6). Categories of impact increased from four to seven given the vulnerability scores, ranging from low to very high impact (Appendix 3.6).

Table 3.3. Scores given by Gulf of Montijo fishers to reflect mangrove vulnerability to threats from human activities. Scores are normalized and sample size refers to the number of interviews used to calculate the vulnerability score.

Human Activity	**Stressors**	**N**	**Vulnerability score**
Agriculture	Non-point source organic pollution	27	1
Aquaculture	Deforestation Nutrients	4	0.75
Centres of population	Littering Deforestation	2	0.5

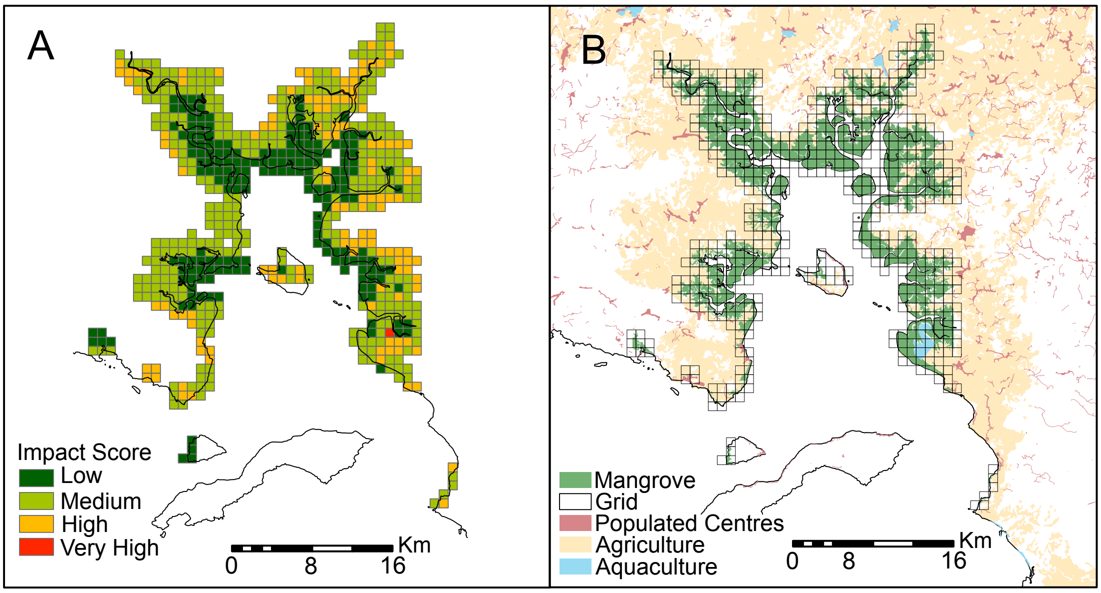

Figure 3.3. The Gulf of Montijo, Panamá. (A) Standard cumulative impact model for human activities per 1km² in the Gulf of Montijo, showing mangroves with low impact in green and high impact in red. (B) The main human activities adjacent to mangroves are agriculture, aquaculture and centres of population.

The weighted model of mangrove state (Figure 3.4) differs from the un-weighted model in revealing that mangroves closer to agriculture fields are more impacted than those adjacent to centres of population (pink coloured polygons in Figure 3.3B). The maximum impact score is reached in only one grid cell where there are aquaculture ponds in the south-east corner of the GoM.

For the un-weighted and weighted impact scores, 30% of cells were classified as low impact, and 70% were medium to very high impact. Of the cells containing impacts, agriculture was the most common, present in 95% of cells, followed by centres of population present in 31% of cells, whilst aquaculture was present in only 2.7% of impact cells in the model. Appendix 3.6 shows the detailed number of cells present in each impact category of the un-weighted and weighted models.

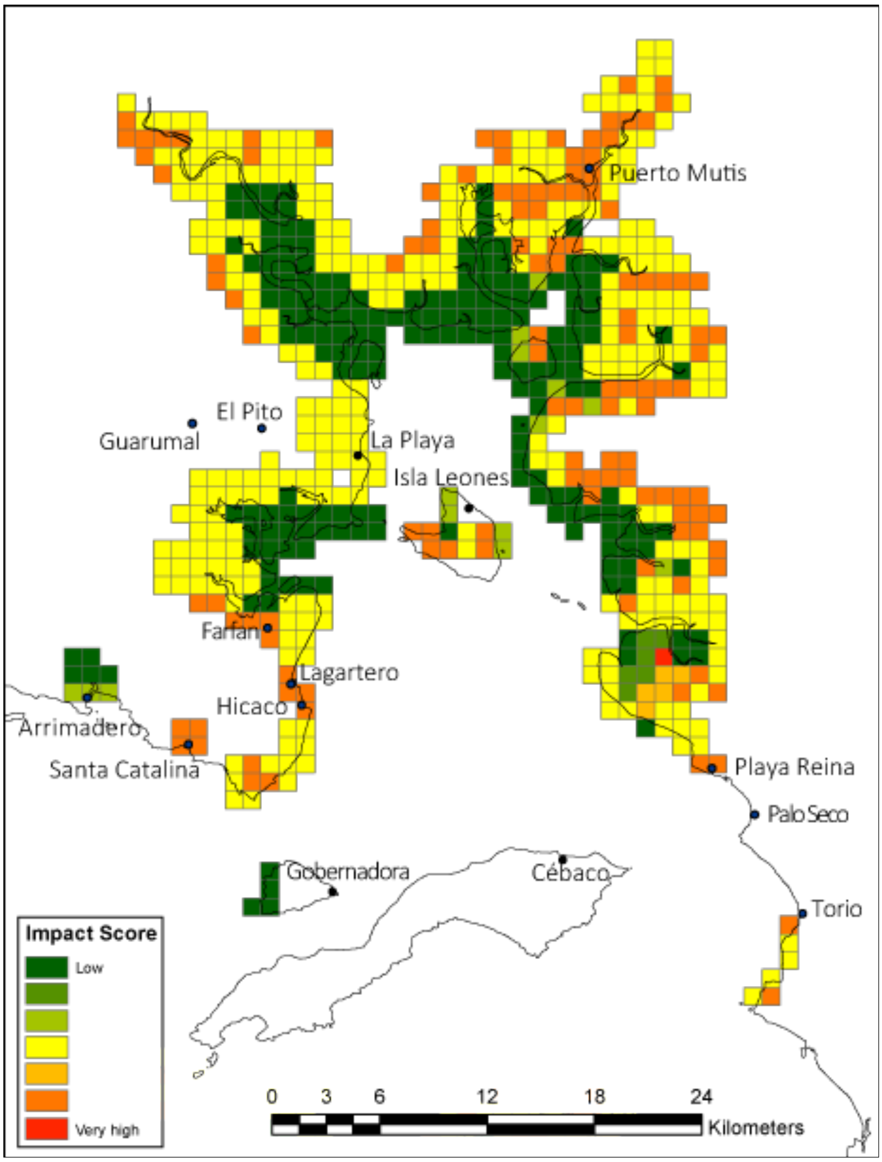

Figure 3.4. The Gulf of Montijo, Panamá. A cumulative impact model for human activities per 1km² in the Gulf of Montijo showing levels of impact from low to high. This model has been weighted based on ecosystem vulnerability scores to different activities (Table 3.3).

3.3.3. Fisheries and mangroves

Government fisheries sampling in the GoM varied widely among villages and categories of species. The landings sampling distribution shows that data sampling effort is low, as most villages across the GoM have large data gaps, with some showing entire years with no sampling (Table 3.4). The village with the highest sampling effort was Palo Seco, and that with the highest sampling in a given year was Hicaco, in 2012.

Table 3.4. Numerical indicators of sampling effort for villages in the Gulf of Montijo (the colours are only to aid visualization, with red showing villages with least sampling, and green showing villages with greatest sampling effort). The number of species-months sampled per village, is shown in parentheses.

Village	Year					Total per village
	2008	2009	2010	2011	2012	
Cebaco	0.0	0.0	0.0	0.3 (49)	0.5 (71)	0.4 (120)
El Pito	0.0	0.0	0.0	0.0	0.2 (31)	0.1 (31)
Gobernadora	0.0	0.0	0.0	0.4 (54)	0.6 (86)	0.5 (140)
Guarumal	0.1 (14)	0.1 (14)	0.3 (51)	0.5 (82)	0.7 (100)	0.9 (261)
Hicaco	0.0	0.0	0.1 (9)	0.6 (90)	**1.0 (153)**	0.8 (252)
Isla leones	0.3 (53)	0.3 (50)	0.0	0.04 (6)	0.0	0.4 (109)
La playa	0.1 (16)	0.0	0.1 (14)	0.0	0.04 (6)	0.1 (36)
Lagartero	0.0	0.0	0.0	0.3 (49)	0.0	0.2 (49)
Farfan	0.0	0.0	0.0	0.0	0.7 (104)	0.3 (104)
Palo Seco	0.5 (70)	0.5 (80)	0.6 (92)	0.2 (32)	0.2 (33)	**1.0 (307)**
Arrimadero	0.0	0.0	0.0	0.0	0.2 (36)	0.1 (36)
Playa Reina	0.1 (20)	0.3 (47)	0.2 (34)	0.2 (23)	0.0	0.4 (124)
Santa Catalina	0.0	0.02 (3)	0.1 (10)	0.1 (9)	0.6 (92)	0.4 (114)
Torio	0.0	0.2 (24)	0.3 (42)	0.2 (25)	0.02 (3)	0.3 (94)

Mangrove species represented more than half (>55%) of landings by number in 9 villages (Figure 3.5), and by weight in 11 villages. For a simple linear regression between the percentage of mangrove species number in landings and mean impact score of mangroves, a non-significant regression equation occurred ($F(1,9)= 3.8$, $p=0.08$, $R^2=0.3$) (Figure 3.6A). The regression between percentage of landings by weight made up of mangrove species and mean impact score was also non-significant ($F(1,9)= 0.4$, $p=0.5$, $R^2=0.04$) (Figure 3.6B).

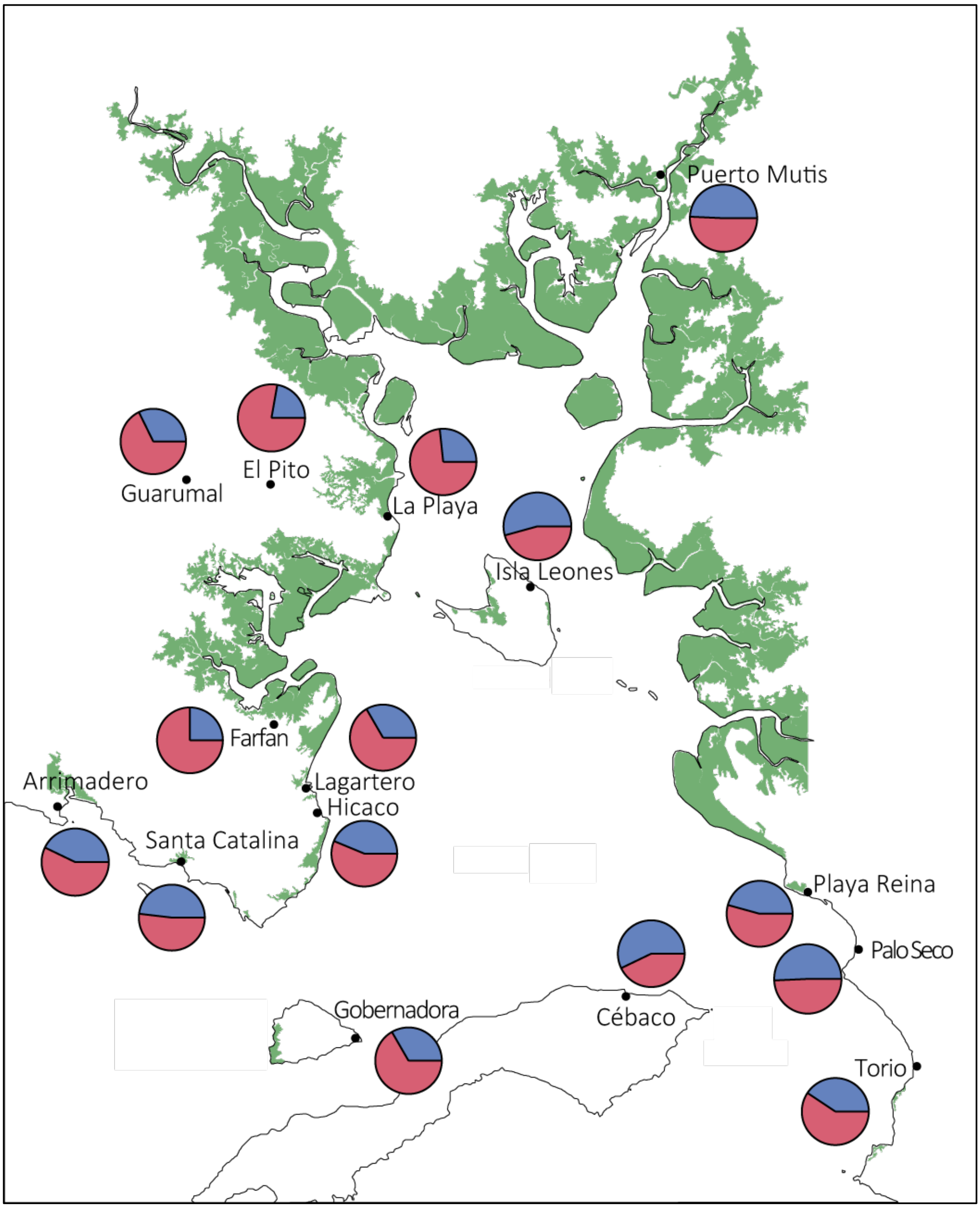

Figure 3.5. Pie charts show the percentage of species number found in total fishery landings of villages from the Gulf of Montijo, Panamá. Pink segments represent the proportion of mangrove-dependent species and blue show non-mangrove dependant species. Mangrove forests are shown in green.

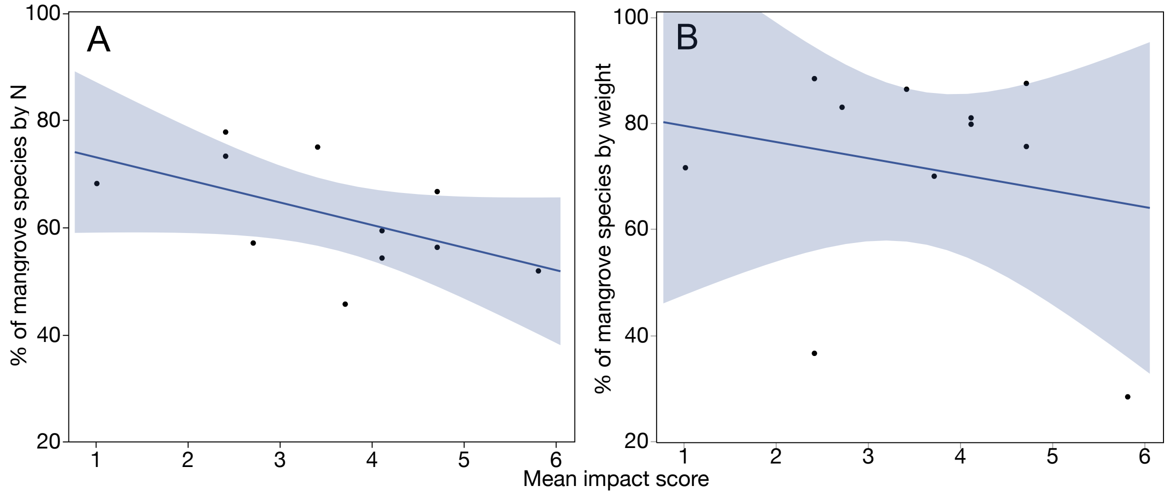

Figure 3.6. Relationship between the mean impact score of mangroves in the vicinity of villages of the Gulf of Montijo and the (A) percentage of number, and (B) weight of mangrove species in fishery landings. Lines represent fitted regressions with 0.95 confidence intervals shaded in blue.

The relationship between distance of villages to the mouth of the GoM and percentage of mangrove-associated species number found in landings was non-significant ($F(1,12)= 3$, $p=0.1$, $R^2=0.2$) (Figure 3.7A). The simple linear regression between distance to the mouth of the GoM and percentage of landings by weight of mangrove species was also non-significant ($F(1,12)= 0.2$, $p=0.6$, $R^2=0.01$) (Figure 3.7B).

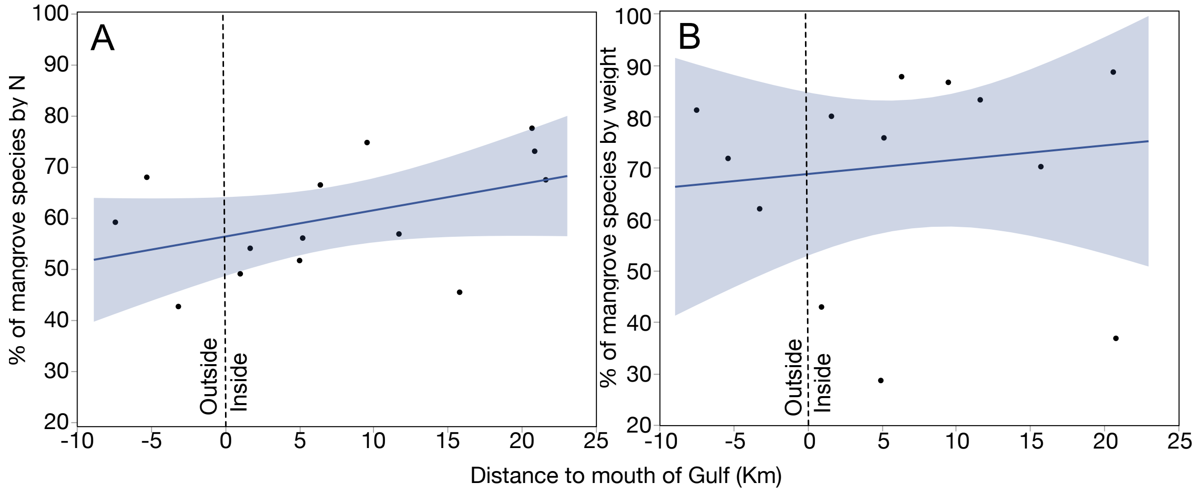

Figure 3.7. Relationship between the distance from villages to the mouth of the Gulf of Montijo, and (A) percentage of mangrove species number and (B) their weight in fishery landings.

The validation of government data with village monitoring data showed that when comparing total estimated landings, government data were an order of magnitude higher on the total landings across all years and mean landings per year, than community monitoring data (Table 3.5).

Table 3.5. Estimated landings of artisanal fisheries in the village of Hicaco, Gulf of Montijo, Panamá. Total estimated landings and mean landings per year are compared between a community monitoring programme, and the government's data.

Data source	Total landings across all years (kg)	Mean landings per year (kg)
Community monitoring	28,704	9,568
Government	203,433	67,811

The mean yearly weight of the total landings from the Hicaco village was used to compare the most important species groups between datasets. Across the three years of sampling (2012-2014) by the community monitoring programme, the most important fishery by year was "revoltura" which is a mixture of low value species (e.g. weakfish, drums, croakers), followed by wahoo, weakfish, catfish, and hammerhead shark (Figure 3.8A). In the government dataset, across the three years of sampling (2010-2012), the most important fishery was also "revoltura", followed by catfish, weakfish, and wahoo (Figure 3.8B).

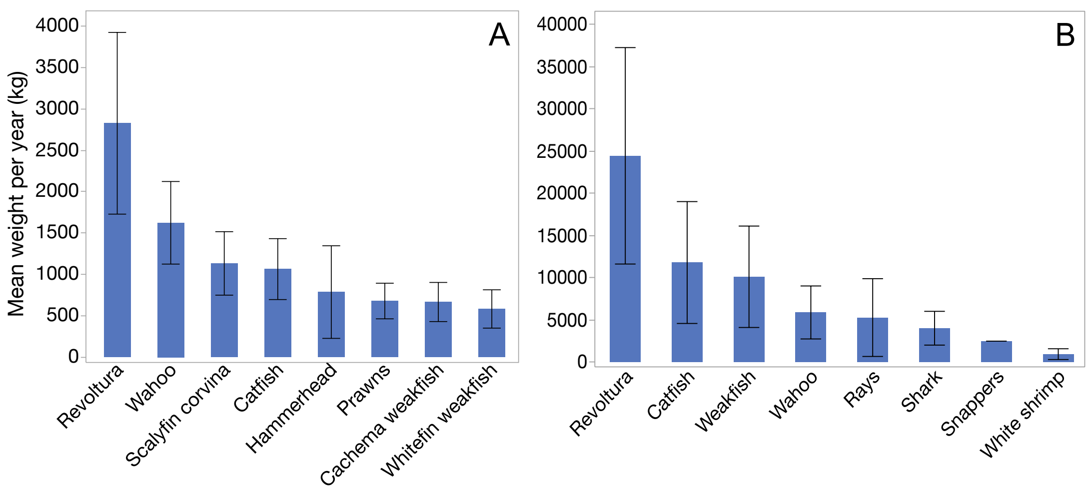

Figure 3.8. Yearly landed weight of top fishery species for the village of Hicaco in two datasets. (A) Community monitoring programme, and (B) government data. Error bars represent the standard error.

3.4. Discussion

The landscape of the GoM is a complex of mangroves and the various human activities exerting influence on them. The impact score model showed that most of the mangroves of the GoM were affected by human activities (70%), with only 30% of cells containing mangroves without any directly adjacent impact. Landings analysis showed mangrove-dependent species were important for the fisheries of the GoM. Out of the 15 villages, mangrove-dependent species contributed to more than 55% of landings by weight for 11 villages, and for 9 villages by number of species. Mangrove condition appeared not to influence fisheries landings. Villages surrounded by mangroves with high impact scores did not have a significantly lower proportion of mangrove-dependent species in landings. These results illustrate not only the limitations of the available data, but the inherent difficulties of studying the link between mangroves and fisheries (Saenger & Funge-Smith 2013).

My study provides a quantitative, spatially referenced assessment of the state of mangroves in the GoM within the context of the larger human landscape around them. Cumulative impact maps of marine ecosystems have proven useful to inform managers on the implementation of ecosystem-based management, marine protected areas and ocean zoning (Halpern et al. 2008; Selkoe et al. 2009; Halpern et al. 2009). Ground-truthing allowed for evaluation of the ability of the impact score model to reliably assess the condition of mangroves in the GoM, in terms of the classification of human activities adjacent to mangroves. Google Earth has been shown to be a powerful, open-access tool for scientists and conservationists to assess the state of an ecosystem (Yu & Gong 2012). In this study, by using more than one source of land use information (i.e. satellite images in Google Earth, GIS layers, interviews, and ground-truthing), I avoided underestimation or misidentification of human impacts.

My analysis shows that most of the mangroves of the GoM are influenced by human activities with agriculture being the most influential activity, present in 95% of the cells, followed by centres of population then aquaculture. The GoM is situated in the second largest agricultural region of Panamá where cattle farming, rice and sugar cane production contribute significantly to the local and national economy (ANAM 2004). However, despite restrictions on mangrove clearing, the area given to rice crops adjacent to mangroves has increased since 2000 (ANAM 2004). This information reflects recent findings which suggest that rice crops are responsible for the fastest rate of mangrove deforestation in Southeast Asia (Richards & Friess 2015).

The damage to mangroves is not limited to direct deforestation. Agricultural crops grown close to mangroves are often ones that need a lot of water (e.g. melon, watermelon and rice), chemical pesticides, and fertilizers (Bach 2007). Agrochemicals and waste products discarded from farms, are transported by rivers and creeks, or are sluiced directly into mangroves and washed into estuaries, killing high numbers of organisms in the water and affecting fish catches (Trejos et al. 2007a; Kaufmann 2012).

In interviews, 100% of fishers said that agriculture posed the most important threat to mangroves and the fish populations they support. Fishers cited specific examples from their own experiences where agriculture has negatively affected mangroves in the GoM directly, because of clearance of mangroves for crops, and indirectly, because livestock husbandry and crop farming can result in toxic quantities of chemicals entering the sea. The scale of this agricultural runoff can be large in the GoM, because fertilization and application of pesticides are not only carried out with ground sprayers, but also from airplanes. Wetlands are known to be routinely contaminated by pesticides and fertilizers on adjacent agricultural areas (Alho & Vieira 1997; Donald et al. 1999; Hill 2003; Hernández-Romero et al. 2004), however the downwind drift from the aerial application has been estimated to be more than four times higher than that produced by ground sprayers (Ware et al. 1969). This might be because aerial fumigation has heightened effects on small streams and ponds, as they are hard to avoid when crops and wetlands share a common boundary (Hill 2003). Studies have shown that the aerial application of pesticides are directly lethal to wetlands wildlife, and that high mortalities are seen not only in fish given high levels of pesticide in water runoff (Hill 2003), but also in birds and mammals (Pimentel 2005). It has been shown that the concentration of pesticides in wetland waters is directly related to precipitation events (Donald et al. 1999), suggesting this may also be the case in the GoM, where large-scale fish mortality is usually observed at the start of the rainy season (Trejos et al. 2007b).

Some studies have shown changes in fish species assemblages following mangrove degradation or clearance (Williamson et al. 1994; Shervette et al. 2007; Shinnaka et al. 2007; Adite et al. 2013). When Adite et al. (2013) examined mangroves of varying degrees of degradation in West Africa, they found that fish species richness and diversity were significantly lower in degraded sites than in restored areas. In

Thailand, Shinnaka et al. (2007) demonstrated that mangrove deforestation had marked effects on fish assemblages, as sites with mangroves had higher numbers of fish species and individuals than sites that had been cleared of mangrove forest. In the present study, I found a lack of relationship between the proportion of mangrove species versus mean impact score, showing that mangrove condition does not influence fish assemblages in terms of mangrove dependency of fishery species. Results found may reflect the uncertainties in this study such as i) lack of accurate information of fishing location, ii) small geographical extent, iii) the debatable relationship between mangroves and fisheries.

The lack of a relationship between distance of villages to the mouth of the GoM and proportion of mangrove-dependent species in landings, might be because landings for some villages may not fully represent species found in the vicinity, but those of more distant fishing grounds, which reflects the need for more accurate information on where fishing takes place. This was a caveat in this study that I aimed to explore further using the community monitoring data, since their data collection sheets included fishing location as names of locally well know fishing areas. However, it was not possible to assign geographical coordinates to these locations as the names provided in the data sheets were subsequently not recognized by consulted fishers. It has been acknowledged that the data collection systems (e.g. catch surveys from selected gear types, vessels, and landing areas) used to characterize small-scale fisheries in the developing world, fall short at doing so because of the dispersed and informal nature of this fisheries (Staples et al. 2004; Mills et al. 2011; Rubio-Cisneros et al. 2016; Tilley & López-Angarita 2016). Mills et al. (2011), estimated catch from landings data of small-scale fisheries of 15 developing countries and found that in all cases national level data were scattered and incomplete, offering a distorted view of the sector. This is probably also the case for Panamá, where small-scale fisheries landings are largely under-reported (Harper et al. 2014) partly due to lack of resources and poor data collection protocols (Mate 2005; Vega et al. 2014).

The small geographic scale of this study relative to the scale of the mobility of mangrove dependent species, may also account for the lack of relationships found between variables explored here. Species movement in mangroves and estuaries responds to many factors including life history (Nagelkerken & Van Der Velde 2002; Mumby et al. 2004; Dorenbosch et al. 2006; Aburto-Oropeza et al. 2009), predatorprey interactions (Dorenbosch et al. 2009; Hammerschlag et al. 2010), feeding habits (Castillo-Rivera et al. 2005; Verweij et al. 2006), and environmental features such as tidal regime, precipitation, salinity, and dissolved oxygen (Barletta et al. 2008; Dantas et al. 2011; Igulu et al. 2014). Distance moved can range from fairly sedentary species (Sheaves 1993) to species that move between habitat types over larger scales (Dorenbosch et al. 2006; Kimirei et al. 2013). Studies have shown that mangrove and estuarine species are highly variable and flexible in their patterns of habitat use, which complicates classifying the dependence of particular species to a given habitat (Blaber et al. 1989; Saenger & Funge-Smith 2013). This flexibility is present across different locations, and has strong temporal variation (e.g. days, years, seasons) (Barletta et al. 2003; Verweij et al. 2006; Barletta et al. 2008; Lacerda et al. 2014). Kimirei et al. (2011), found high spatio-temporal variability in the ontogenetic shifts of four fish species associated mangroves and seagrass in Tanzania. Accordingly, since fish composition varies greatly in temporal and spatial scales following the complex interaction of multiple factors (Kimirei et al. 2011; Sheaves 2016), it is difficult to determine if changes respond to the natural variability of assemblages or to the condition of the ecosystem (Sheaves et al. 2012).

Notwithstanding the mentioned limitations and uncertainties, this study is important as a first step in assessing the fisheries in the Gulf in terms of mangrove dependency. In the GoM, mangrove species dominated more than half of the landings for many villages (Figure 3.5), and the majority of fishers recognized the importance of mangroves for their livelihoods, as seen elsewhere (MacKenzie 2001; Walters et al. 2008; Hussain & Badola 2010; UNEP 2014). Given the shortcomings of current fisheries data collection protocols in small-scale fisheries, the contributions of this sector to food security, local livelihoods, and poverty alleviation are likely to be undervalued in Panamá, as it is often the case in developing nations (Mills et al. 2011). One way to tackle this is the collection of fisheries data with household socio-economic data, as this provides a robust insight into the societal role of fisheries and ultimately contributes in a meaningful way to policy development (Mills et al. 2011).

The map of human impacts for the GoM showed that inland boundaries of mangroves were likely to be influenced by one or more human activities simultaneously, with cumulative consequences. Hence the protected area designation of the GoM, as a Wetland of International Importance, is failing to adequately protect against indirect influences of peripheral land use activities, causing negative impacts on mangrove integrity and associated biodiversity. My results highlight often overlooked impacts on mangroves from adjacent land use, and call for management strategies to regulate human impacts threatening mangroves resources, especially for the effects of fertilizers and pesticides on fishery species. For example, given that wetlands in agricultural landscapes are exposed to high levels of pesticides (Donald et al. 1999) fumigation should be regulated. This could be the first step towards a spatial planning approach that integrates management and legislation of multiple user groups and stakeholders (Crowder & Norse 2008) which is crucial to reduce the impacts that human activities cause on mangroves and fish stocks. My findings illustrate the need to improve fisheries data collection as it could benefit fishers by allowing management to be tailored to the reality of the GoM, given the observed high reliance of local communities on mangrove-dependent fishery species.

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Chapter 4. Winners and losers in area-based fisheries management: A case study of small-scale fisheries in the Colombian Pacific.

Abstract

The Pacific coast of Colombia has some of the most extensive mangrove forests in South America. It is an isolated region and one of the country's poorest, where coastal communities rely on fishing as a main source of animal protein and income. In an attempt to reverse declining trends of fisheries resources, in 2008, an Exclusive Zone of Artisanal Fishing (ZEPA) that banned gill nets, was established by stakeholders in the Northern Chocó region. In this chapter I investigate the effects of this area-based management on fisheries productivity and catch composition through a case study in the Northern Chocó. Fishery landings data from 2010 to 2013 are compared to those of a neighbouring area with no fisheries management. Catch per unit effort, mean weight landed, and number of landed individuals, were calculated for mangrove and non-mangrove associated species. Results suggest that management significantly improved fisheries productivity across gear types and time, with mean catch per unit effort increasing by 50% in the ZEPA within 3 years. Fisheries here focused on offshore resources with 61% more fishing trips associated with motorized boats than in the unmanaged region, where fishing was predominantly in mangroves and close to the coast. This suggests that fisheries management, and in particular territorial use rights in fisheries, has reduced pressure on mangrove resources and enhanced small-scale fisheries productivity in the ZEPA. However, findings also show the ZEPA might have caused a displacement of fishing effort by excluding industrial trawlers, who concentrated their activity in neighbouring areas.

4.1. Introduction

Fisheries are important to the national economies of many developing countries, through contributions to food security and supply, employment, livelihoods, and poverty alleviation (Finegold 2009; Béné 2009; Mills et al. 2011). However, as fisheries are declining worldwide all these are threatened by that loss (Pauly & Zeller 2015). Diverse strategies have been proposed to reverse fisheries decline, and include the creation of protected areas, regulation of fishing activity (e.g. total allowable catch, gear restrictions, temporal restrictions), and protection of essential fish habitat (Gell & Roberts 2003). In the tropics, mangrove habitat offers critical refuge and food to associated fish and invertebrate species (Nagelkerken et al. 2001; Aburto-Oropeza et al. 2008) before they migrate elsewhere (Mumby et al. 2004). Consequently, mangrove protection may serve to safeguard or restore fisheries productivity (Aburto-Oropeza et al. 2008).

Several studies have investigated links between fisheries productivity and physical variables of mangroves, such as forest area and perimeter. Turner (1977) and Martosubroto & Naamin (1977) provided the first evidence that higher shrimp production correlated with better mangrove cover, which was corroborated by Yañez-Arancibia et al. (1985) and Pauly & Ingles (1986). Since then others have argued that as links found between mangroves and fish abundance are based upon correlations, they do not necessarily imply causality (Baran 1999), and that factors related to mangrove cover, such as occurrence of extensive shallow seas, intertidal area, tidal creeks, organic matter, and length of coastline, may underlie the high auto-correlation of variables (Baran & Hambrey 1998). In 2005, Manson et al. reported that in Australia mangrove perimeter and area accounted for most variation in catch per unit effort of mangrove-related species. In the Gulf of California, Aburto-Oropeza et al. (2008) found a positive relationship between mangrove-related fish and crab species landings and the area of the mangrove-water fringe. Furthermore, a study of the Wider Caribbean showed that mangrove habitats serve to enhance reef fish abundances at a regional level (Serafy et al. 2015).

Mangroves in Colombia host a great diversity of important fishery species such as cockles, prawns, shrimp, crabs and fish such as catfish, snook, snapper and tarpon (Villalba 2005). Mangroves are abundant on the Pacific coastline around its extensive sheltered estuaries, bays and rivers (Prahl et al. 1990). By contrast, on the Atlantic coast, wave climate is often too energetic for mangroves (Álvarez-León & Polanía 1996). On the Pacific coast, mangroves occur in a broad band in the south, becoming patchy in the north where there are fewer big rivers (Prahl 1989).

The Pacific coast of Colombia has remained isolated from economic growth in the rest of the country due to weak presence of government institutions and a lack of infrastructure development and hence, accessibility (Camacho et al. 2000; Díaz & Galeano 2016). Consequently, the Pacific region is mostly jungle with small, scattered fishing villages inhabited by indigenous tribes and Afro-descendants (Prahl et al. 1990). Economic activities in these isolated communities are limited to fishing and small-scale agriculture (Blanco et al. 2011). Fishing is mainly for subsistence with commercial activity limited due to restricted possibilities for export (Díaz & Galeano 2016).

The Colombian Pacific fisheries are highly productive, sustaining 80% of Colombia's total fish catch (Díaz et al. 2011), however since 1990s catch levels have been steadily decreasing (Lindop et al. 2015). It is now known that many fisheries have exceeded their maximum sustainable yield and that resources are overexploited (Díaz et al. 2011). This was highlighted by the overexploitation of the white shrimp (*Litopenaeus occidentalis*) and titi shrimp (*Xiphopenaeus riveti*) fisheries in the Pacific since 1990s (Rueda et al. 2001). In response industrial fishing fleets were drastically reduced by the diminishing resources (Baos Estupiñán & Zapata 2011). This situation led to further encroachment from industrial fishers into coastal waters to boost their catches (García 2010) which in turn fuelled self-mobilised, artisanal fishers of the Northern Chocó region to create the *Exclusive Zone of Artisanal Fishing* (ZEPA) in 2008 (Vieira et al. 2016). This area excluded industrial fishing within 4.6 km of the coast, and incorporated management measures such as the prohibition or gillnets and beach seines to allow the recovery of overfished populations (Vieira et al. 2016).

The ZEPA is an example of "territorial use rights in fisheries" (TURFs), a term coined by Francis Christy in 1982, to refer to the allocation of rights to use all or part of a resource in a particular geographic space (Christy 1982). The "sea tenure" of TURFs is not a full ownership right to resources, but instead access rights are granted and actual ownership resides with the nation (Wilen et al. 2012). In this case study, I investigate how a TURFs style of fisheries management has affected catch composition by comparing landings data from the ZEPA, with a comparable unmanaged area. Additionally, given the need for more information on the contribution of mangroves to subsistence fisheries (Saenger & Funge-Smith 2013), I examine the importance of mangrove-associated species in the small-scale fisheries of Northern Chocó between 2010 and 2013.

4.2. Methods

4.2.1 Study Site

This study took place in the Northern Chocó region of Colombia's Pacific coast (Figure 4.1), a global biodiversity hotspot with high levels of endemism in flora and fauna (Myers et al. 2000; Díaz & Gast 2009). The area has some of the world's highest rainfall with figures of 5000 to 7700 mm per year, and an average relative humidity of ~89% (Díaz & Galeano 2016). Normally a rainy season occurs between May – November with dryer weather from December – April, but this varies according to the El Niño–Southern Oscillation (Wang & Fiedler 2006). The coastline is characterized by rocky hills and precipices and receives input from many rivers. Large estuaries occur in Juradó, Cupicá, Tribugá and Nuquí, where the main mangrove forests occur, and these cover approximately 4978 ha. The population of ~19,000, are predominantly Afro- descendants and indigenous people from the "embera" culture (Camacho et al. 2000). The most common economic activities are cattle farming, agriculture and fishing (Blanco et al. 2011). However, more than half the population lives in poverty, without basic needs (Camacho et al. 2000; Blanco et al. 2011).

The region has two distinct fishery management zones: the Gulf of Tribugá (hereafter referred to as Tribugá) where there are no fishing regulations in place; and the Exclusive Zone of Artisanal Fishing (ZEPA) (García 2010) where fisheries management exists. In Tribugá there is one protected area called Utría National Park, which was declared in 1986 (García 2010) (Figure 4.1). However, since this Park does not have distinct regulations to restrict fishing activities, I treat it as part as of Tribugá. In the ZEPA, fishing regulations prohibit the use of gillnets and only allow hand lines and long lines (Vieira et al. 2016).

4.2.2 Data sampling

To compare the fisheries of the ZEPA and Tribugá, I examined data collected between March 2010 and September 2013 from 16 coastal communities in the two areas by the Marviva Foundation (Marviva). This organization trained local people to collect fishery information using surveys designed for a national information system of landings (Neira et al. 2016). Sites where the information was collected were chosen within all landing areas such that all fishing gears used would be considered. Sampling frequency varied according to budget availability and local conditions. In general, catch and effort data were recorded 3-4 times per week. Sampling was uniform throughout the year apart from where local religious holidays or restrictive inclement weather created gaps. The information I examined from the Marviva dataset was: catch weight and number, species landed, type of boat used, number of fishers, name of boat captain, trip length, fishing gear, and fishing location. Data recorders for Marviva mapped the location of fishing grounds (i.e. sites where people regularly fish) by accompanying fishers on trips and recording a suitable single GPS point for different localities.

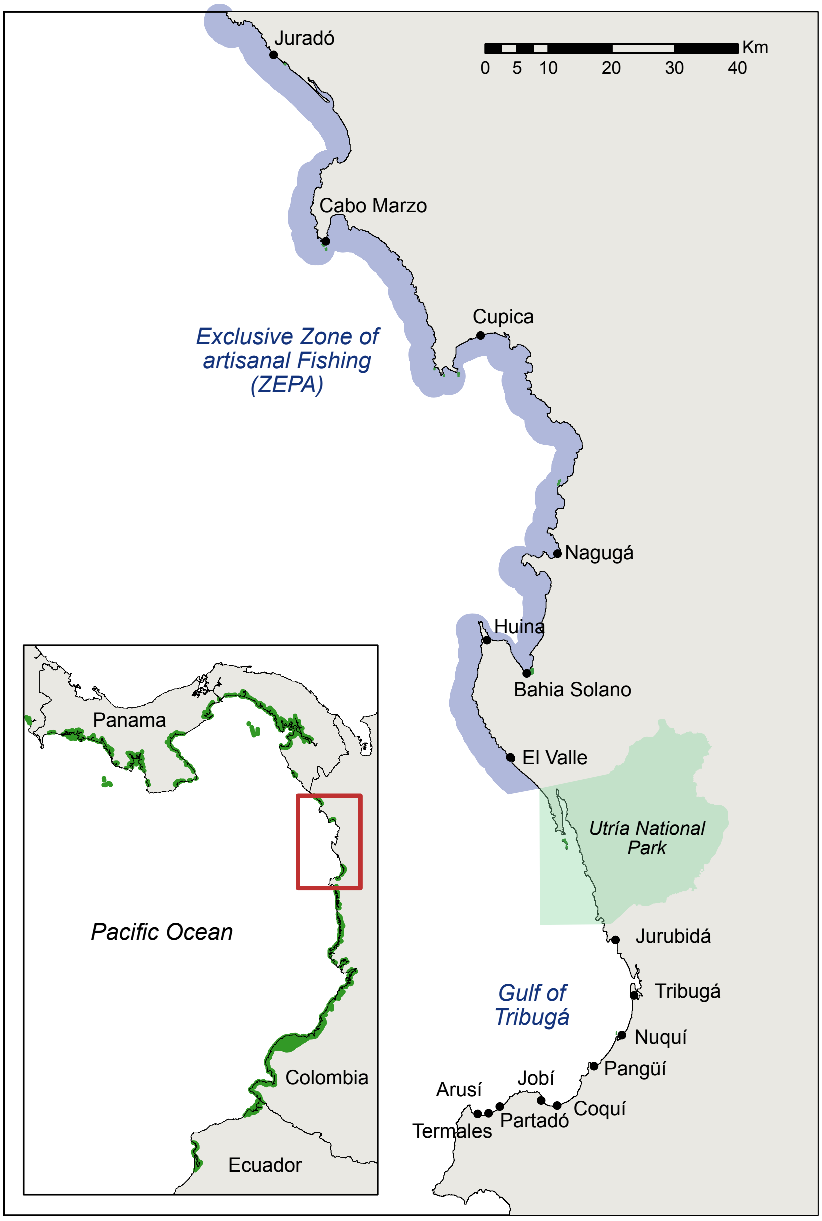

Figure 4.1. Study site location in the Northern Chocó region on the Pacific coast of Colombia (red square, inset). Solid black circles represent artisanal fisheries landing sites included in this study. Blue shading represents the Exclusive Zone of Artisanal Fishing (ZEPA). Mangroves are shown in green.

4.2.3 Characterizing artisanal fisheries of Northern Chocó

Landings data for the ZEPA and Tribugá were analysed to characterise fisheries according to gear type, species composition and diversity, and fishing location. Temporal change in landings over the study period was then calculated for these fisheries. Geographic locations of fishing grounds were mapped in ArcGIS 10.3.1 as point data, and interpolated to create a spatial density map of the fishing grounds in each area.

Relative importance of species and gear types were calculated in terms of total weight landed. Catch per unit effort (CPUE) was used as a measure of relative fish abundance across time, and to compare gear types and management zones. Catch was summed in kilograms and fishing effort was calculated as fisher days whereby the number of days fishing was multiplied by the number of fishers on board the vessel.

To explore catch species diversity, total and mean number of species landed per gear were calculated by fishing ground, and then compared among gear types using Wilcoxon tests. The relationship between number of gears employed in a fishing ground and the total number of species present in landings was explored further by running a non-parametric Spearman's rho correlation.

To understand spatial patterns in the fisheries I calculated the distance from each fishing ground to the coast using ArcGIS 10.3.1. To do this I overlaid the map of points of fishing grounds over the map of the coastline of Northern Chocó, and used the spatial analyst extension in ArcGIS to calculate the most direct distance from each fishing ground to the closest point in the coastline. Mean distances from fishing grounds to the coast were compared between the ZEPA and Tribugá using a Kruskal-Wallis test. Proximity of fishing grounds to the coast was also calculated per boat type. The latter were classified as motorized or non-motorized with motorized boats also sub-classified according to engine sizes of: <16HP, 16-40HP, and >41HP. To determine the relative importance of each boat type for the fisheries, I calculated the proportion of each boat type in the total number of trips recorded for each area. Temporal trajectories of fisheries from the ZEPA and Tribugá were estimated by comparing annual CPUE (kg/fisher·day) of finfish species by summing weight landed per month and dividing it for the monthly effort values (kg/fisher·day) for each of the years. To determine if CPUE changes over time were significant, CPUE was log transformed and years were compared using an ANOVA for the ZEPA and Tribugá independently. All variables were tested for normality. Statistical analyses were performed in JMP ver. 13.

4.2.4 Exploring the influence of mangroves on artisanal fisheries

To analyse the proportion of mangrove dependent species in landings I defined mangrove dependence following the criteria of Blaber (1989) as: "*Species for which estuaries or similar habitats are the principal environment for at least one part of their life cycle*". Given there is incomplete knowledge of the life cycle of many eastern Pacific species I attributed mangrove dependence according to recognised information for similar species as described in Appendix 4.1.

Temporal trajectories of relative abundance of mangrove dependent and non-mangrove dependent fish species were compared between Tribugá and the ZEPA using the proxy measure of CPUE. CPUE was calculated by summing the weight of landings per month sampled and dividing it by the monthly effort (fisher·days) for each of the years. To explore the role of mangrove species in the fisheries of Northern Chocó, the independent variables of management zone, boat size, and species dependence on mangroves, were used to predict the outcome variables of: number of individual fish landed per trip, weight of landings per trip, and CPUE. Boat size was used as proxy variable for access to financial capital, as more wealthy fishers or owners will have access to powered boats in comparison with less wealthy fishers who will use man-powered boats. Boat size was grouped into non-motorized and motorized. Distance from fishing grounds to the coast was not included in this analysis given its auto-correlation with boat type.

A two-way analysis of variance was performed where possible and separately for the ZEPA and Tribugá. Weight landed per trip and CPUE were transformed using the Log function to fulfil the parametric requirements of the test. The influence of boat size and mangrove dependence on the mean of fish and shellfish landed per trip were analysed using a two-way mixed factorial ANOVA. This same method was used to examine the influence of boat size and mangrove dependence on CPUE.

The number of individuals landed per boat could not be transformed to achieve a normal distribution for a two-way parametric factorial analysis, so instead one-way analyses were conducted. Mean number of individual fish and shellfish landed per trip was compared between categories of boat types using a Kruskal-Wallis test.

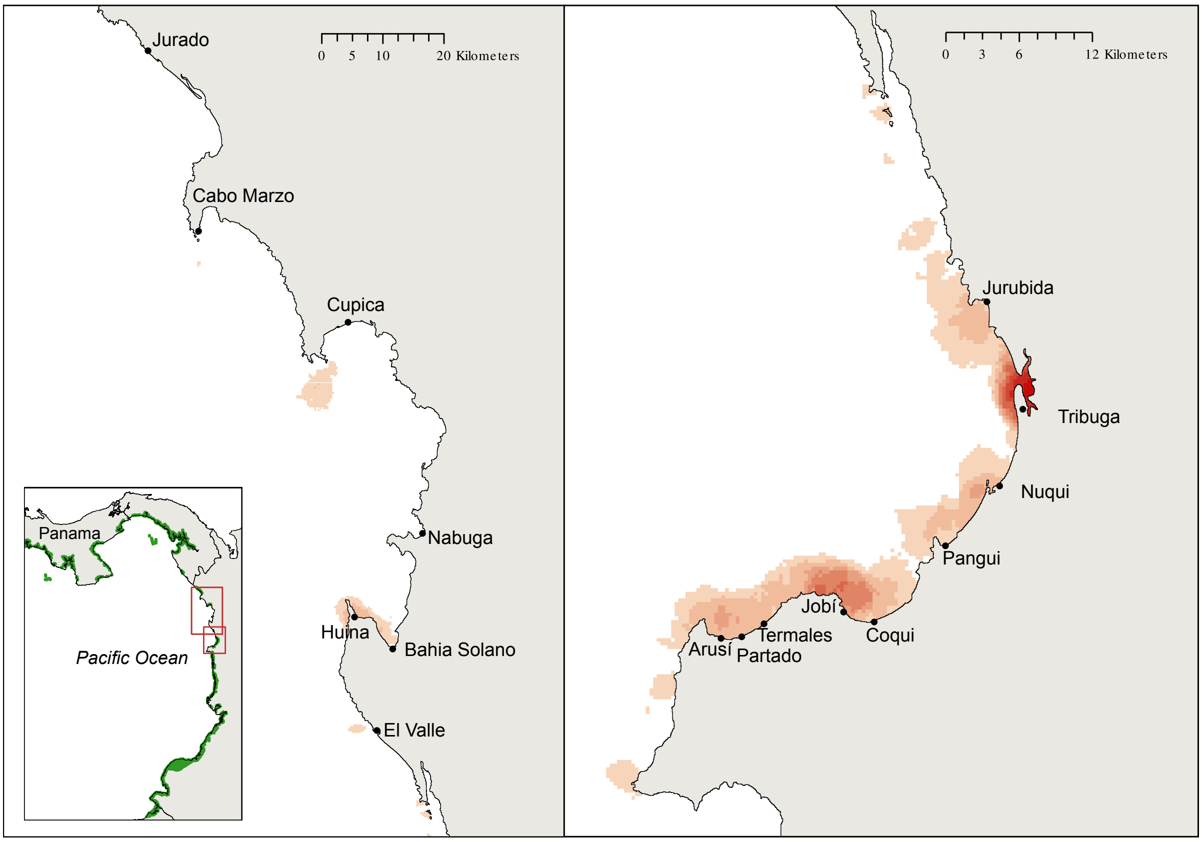

Figure 4.2. Density map of fishing grounds in the Exclusive Zone of Artisanal Fishing (ZEPA) and the Gulf of Tribugá, showing the distribution of fishing sites in the area. Dark red areas represent high fishing ground density and light red represents low density.

4.3. Results

4.3.1 Fisheries of Northern Chocó

Fisheries landings data were recorded in seven fishing communities in the ZEPA and nine in Tribugá. 270 fishing grounds were used in total with 179 located in Tribugá, and 91 in the ZEPA. These were more densely concentrated in Tribugá than in the ZEPA, where the distribution was predominantly uniform (Figure 4.2). The maximum distance fishers ventured from the coast was 13.8 km. Data were recorded for 6,054 fishing trips within the ZEPA and 30,394 within Tribugá. In the ZEPA, mean monthly fishing effort for 3 years of sampling was 557 fisher-days, versus 1,832 for Tribugá across 4 years of sampling. The mean annual fishing effort for the ZEPA was 6,128 fisher-days compared to 21,071 for Tribugá. In the last population census in 2005, 8,475 people lived in the ZEPA and 7,089 Tribugá (Appendix 4.2) and experts familiar with the region consider these numbers still reflect the current demography (JM Díaz, *pers com,* Marviva Science Director).

In Tribugá, 116 species were identified in landings, with 80 species from the ZEPA. This combined value increased to a total of 284 when including species with unknown scientific nomenclature. Only identified species were included in the analysis. Fifty-five percent of species were classified as mangrove dependent. In the ZEPA, when total weight landed by species was calculated, *Thunnus albacares* was most the most important species with *Sphyraena ensis* for Tribugá (Figure 4.3). By this metric, other species that were important to both areas were: *Brotula clarkae*, *Caranx caninus, Caranx caballus, Seriola rivoliana*, *Lutjanus guttatus, Lutjanus peru,* and *Scomberomorus sierra* (Figure 4.3).

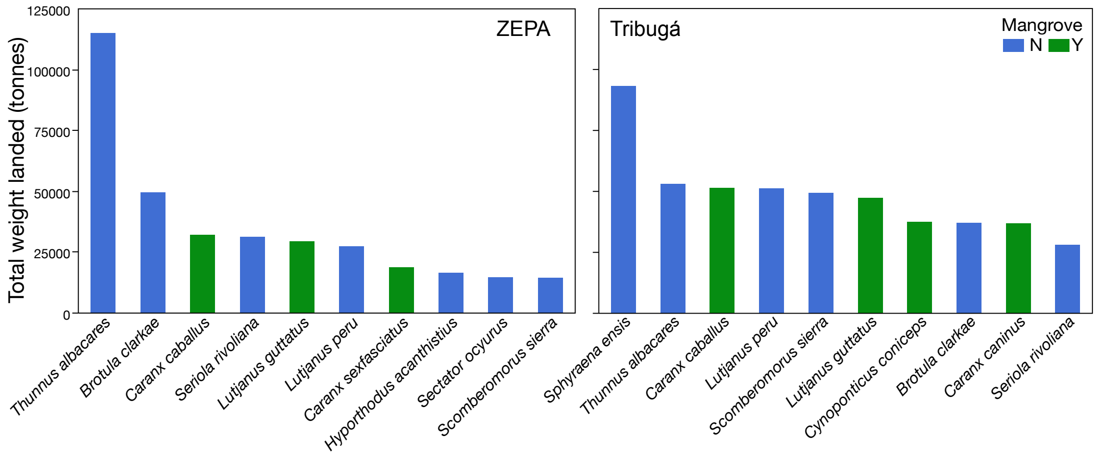

Figure 4.3. Top 10 species by weight landed (tonnes) between 2010 and 2013 in the Exclusive Zone of Artisanal Fishing (ZEPA) and the Gulf of Tribugá. Green bars represent mangrove associated species and blue bars represent species not associated with mangroves.

For both the ZEPA and Tribugá, the main gear types by mean annual weight of landings were: hand lines, followed by long lines, gillnets, spear guns, manual collection for molluscs and beach seine (Table 4.1). Landings associated with cast net were only present in Tribugá. However, the relative importance of gear types changes when it is calculated using catch per unit effort, with beach seine showing the highest yields in both regions, followed by long line (Table 4.1). Mean CPUE was significantly different by gear type for Tribugá (Wilcoxon, χ^2=922.7, DF=6, P=<0.001) and the ZEPA (Wilcoxon, χ^2=122.8, DF=5, P=<0.001). The Tukey-Kramer test was used for post-hoc comparisons (Appendix 4.3). The most frequently used gear type was hand line, used on 71% of trips in Tribugá and 69% in the ZEPA. In Tribugá, gill nets were the second most important gear type, used on 18% of trips, followed by long line used on 9%. In the ZEPA, long lines were the second most important gear, used on 17% of trips, followed by gillnets which were used on 13%.

Table 4.1. Fishing gears used in Northern Chocó region between 2010 and 2013, showing mean weight ± standard error of annual catch and catch per unit effort.

Fishing gear	Mean annual landings (kg) ± SE		Mean catch per unit effort ($kg \cdot fisher^{-1} \cdot day^{-1}$) ± SE	
	ZEPA	Tribugá	ZEPA	Tribugá
Hand line	107,084 ± 54.1	122,726 ± 60.9	35.8 ± 0.5	17.5 ± 0.1
Long line	431,27 ± 26.4	34,138 ± 35.8	42.8 ± 1.2	29.4 ± 0.8
Gillnets	23,058 ± 32.1	45,190 ± 36.94	39.5 ± 0.1	18.3 ± 0.3
Spear gun	1,137 ± 11.9	717 ± 4.3	33.7 ± 5.0	19.4 ± 2.1
Manual collection	5.5 ± 1.1	1,057 ± 11.8	1.8 ± 0.5	5.4 ± 0.7
Beach seine	935 ± 12.4	481 ± 7.1	48.2 ± 6.4	37.8 ± 6.8
Cast net	-	32 ± 1.3	-	5.8 ± 1.0

Mean CPUEs for hand line, gillnet and long line were consistently higher in the ZEPA than Tribugá (Figure 4.4) (hand line: Kruskal-Wallis, $Z=46.70$, $P<0.001$; gill net: Kruskal-Wallis, $Z=28.91$, $P<0.001$; long line: Kruskal-Wallis, $Z=19.43$, $P<0.001$).

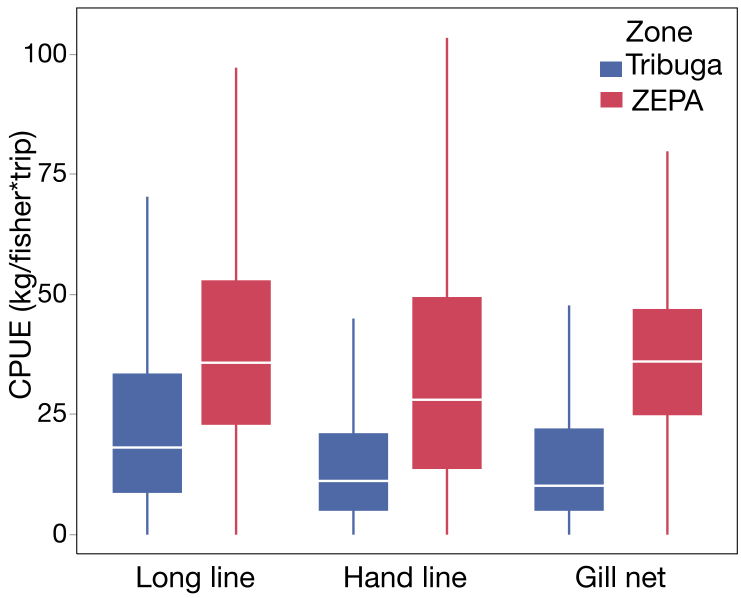

Figure 4.4. Catch per Unit Effort (CPUE) for the principal gear types in the Exclusive Zone of Artisanal Fishing (ZEPA) and the Gulf of Tribugá between 2010 and 2013. In the boxplots, the

line inside the box represents the median, and vertical lines indicate variability inside the upper and lower quartile.

Across all fishing grounds, hand line and gill net landings had significantly higher diversity of species than all other gear types (Figure 4.5A) (Wilcoxon, Z = 2.74, P=0.006,), but these did not differ significantly themselves (Wilcoxon, Z =-0.088, P=0.93). Mean (± SE) number of species landed per trip by gill nets was 20.6 ± 1.32, and for hand line was 19.7 ± 1.02 (Figure 4.5A). Hand lines caught the most species in total with 134 species, followed by gill nets with 126, long lines with 111, spear with 46, seine net with 21, cast net with 12, and manual collection targeting 2 species. The relationship between number of gears employed by fishing ground and the total number of species landed was positive and relatively strong (Spearman's ρ=0.73, p=0.001) (Figure 4.5B).

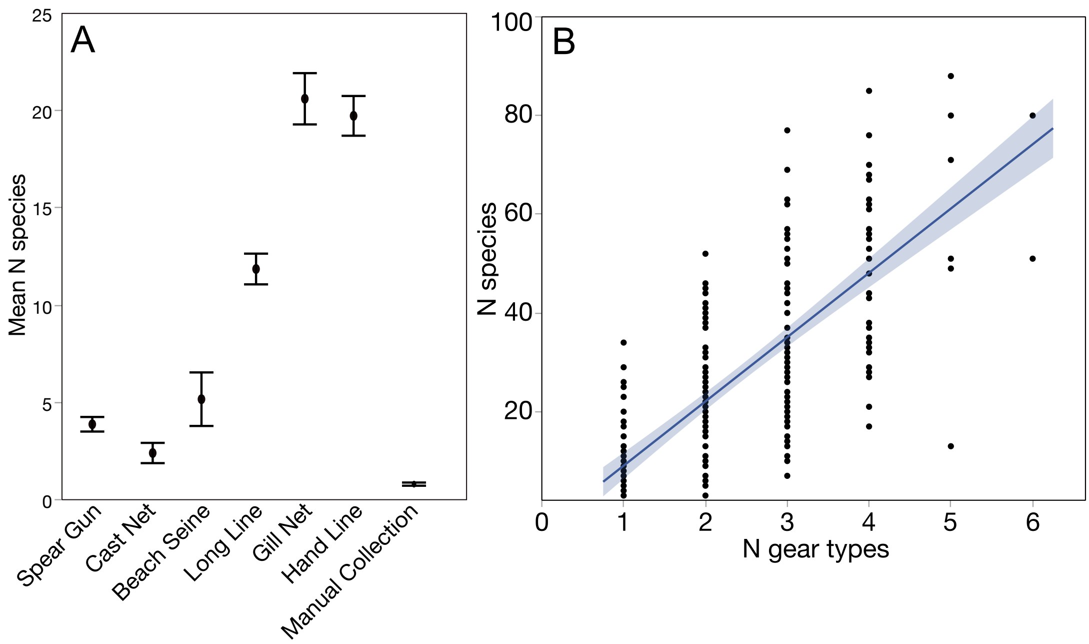

Figure 4.5. (A) Mean number of species (± SE) landed by different gear types in Northern Chocó between 2010 and 2013. (B) Relationship between the mean number of species landed in fishing grounds and the mean number of gears used.

In Tribugá, most fishing (75%) was conducted with non-motorized vessels, whereas in the ZEPA motorized boats accounted for 86% of boat trips of which 88% were 16HP or less (Table 4.2).

Table 4.2. Proportion of fishing trips associated with different types of boats in the fisheries of the Exclusive Zone of Artisanal Fishing (ZEPA) and the Gulf of Tribugá between 2010 and 2013.

Boat Type	Tribugá	ZEPA
Non motorized	75%	14%
Motorized	25%	86%
<16 HP	92%	88%
16 - 40 HP	5.5%	8.7%
41 - 75 HP	2.2%	2.4%

Distance from the coast to fishing grounds was significantly different for the ZEPA and Tribugá (Kruskal-Wallis, $Z = 5.315$, $P<0.001$), where average figures were 2.65 km ± 2.9 within the ZEPA and 0.95 km ± 1.3 in Tribugá (Figure 4.6A), with a maximum distance of 13.8 km and 7.9km respectively. In Tribugá, 32 fishing grounds were located inside mangroves with none for the ZEPA. Figure 4.6B shows that mean distance travelled increases with boat size.

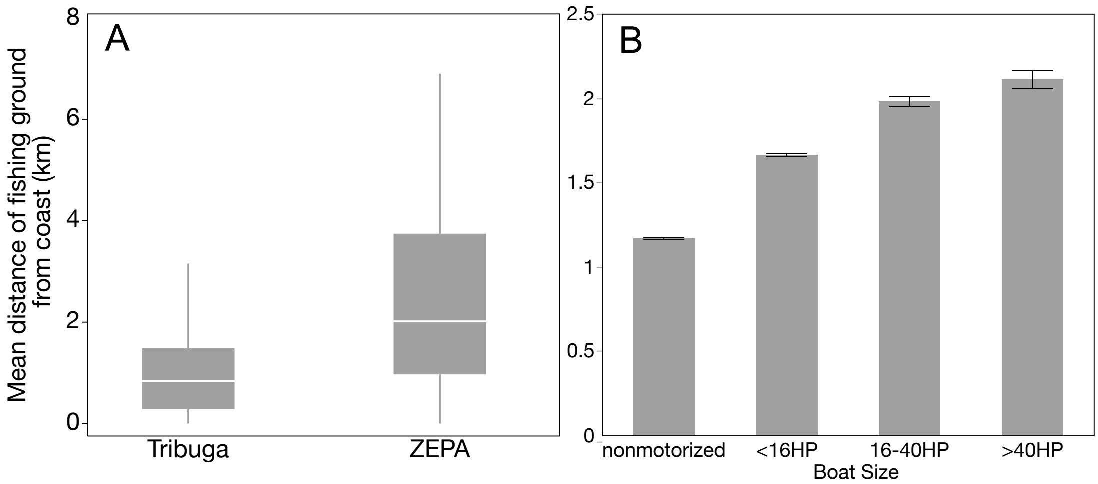

Figure 4.6. (A) Distance from fishing grounds to the coast for the Exclusive Zone of Artisanal Fishing (ZEPA) and the Gulf of Tribugá, and (B) proximity of fishing grounds to the coast in relation to boat size in Northern Chocó region between 2010 and 2013. In the boxplots, the line inside the box represents the median, and vertical lines indicate variability inside the upper and lower quartiles. Error bars in B represent the SE of the mean.

Mean annual CPUE (kg/fisher·day) indicate that the ZEPA and Tribugá both started with fisheries of similar productivity but have diverged over time, with CPUE increasing 50% in 3 consecutive years in the ZEPA after management was introduced in 2008, compared with a stable trend following an initial decline across the 4 years of sampling in Tribugá (Figure 4.7). In the ZEPA, mean CPUE was significantly different between years sampled ($F(2, 30)=24.12$, $P<.0001$), and this was also the case for Tribugá (Kruskal-Wallis, $Z=43.28$, $P<0.001$).

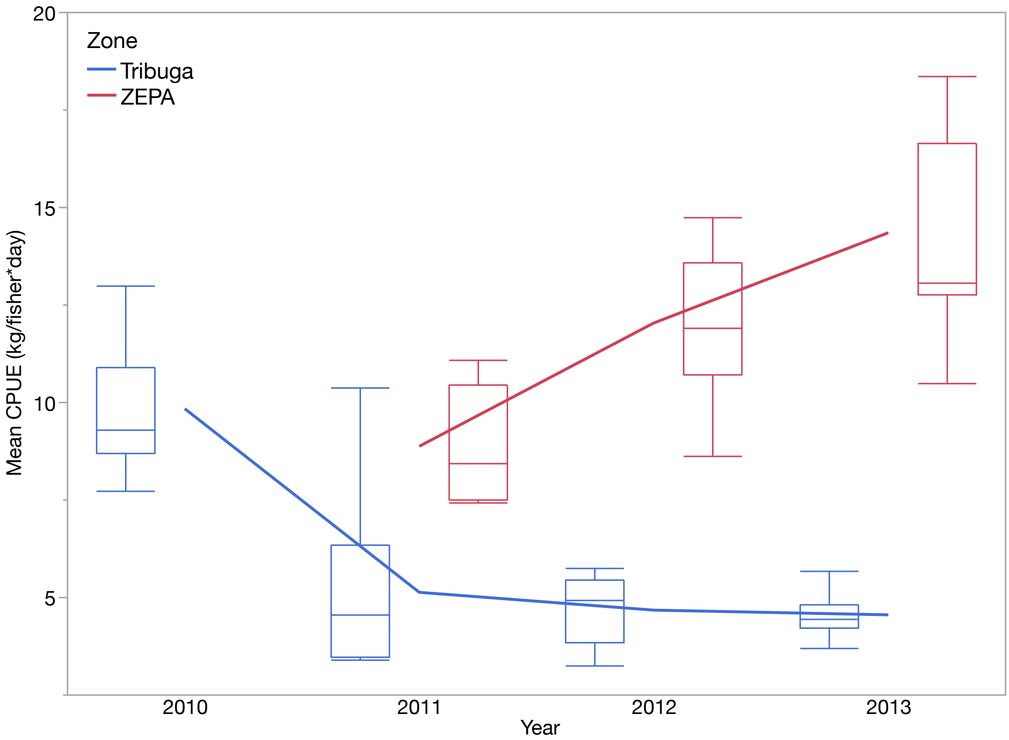

Figure 4.7. Temporal changes in Catch per Unit Effort (CPUE) in the Exclusive Zone of Artisanal Fishing (ZEPA) and the Gulf of Tribugá, Colombia between 2010 and 2013. In the boxplots, the line inside the box represents the median, and vertical lines indicate the range of the data inside the upper and lower quartiles.

4.3.2 The role of mangroves in artisanal fisheries

Mangrove and non-mangrove species CPUE over time was higher in the ZEPA than in Tribugá. CPUE of non-mangrove species was higher than mangrove species (Figure 4.8).

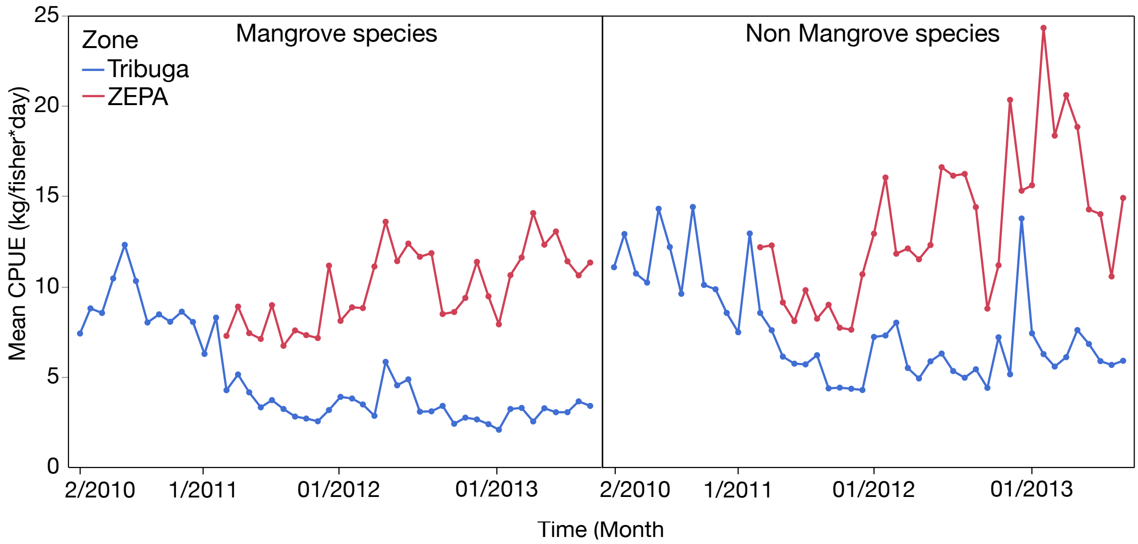

Figure 4.8. Temporal changes in mean Catch per Unit Effort (CPUE) for mangrove and non-mangrove associated species, showing separate trend lines for the Exclusive Zone of Artisanal Fishing (ZEPA) and the Gulf of Tribugá between 2010 and 2013.

The effect of mangrove dependence and boat type on weight landed and CPUE were significant for both zones (Table 4.3). Mean landings weight and CPUE were higher for the ZEPA than Tribugá in all cases (Table 4.4) (Figure 4.9). Motorised boats exhibited higher mean trip landings and CPUE than non-motorised crafts in both zones (Table 4.4). Mangrove-associated species landings showed lower weight and CPUE than non-mangrove species (Figure 4.9).

Table 4.3. Results from two-way factorial ANOVAs for fisheries landings data from the Exclusive Zone of Artisanal Fishing (ZEPA) and the Gulf of Tribugá taken between 2010 and 2013. Parameters and significance values for main effects are detailed.

Zone	Dependent variable	Independent variable		
		Mangrove	Boat type	Interaction
ZEPA	Weight of fish landed	$F(3, 8692)=143.5$, $P<0.001^*$	$F(3, 8692)=1464$, $P<0.001^*$	$F(3, 8692)=0.07$, $P=0.7$
	CPUE	$F(3, 8688)=241.7$, $P<0.001^*$	$F(3, 8688)=55.4$, $P<0.001^*$	$F(3, 8688)=9.1$, $P<0.01^*$
Tribugá	Weight of fish landed	$F(3, 43285)=684.6$, $P<0.001^*$	$F(3, 43285)=$ 8858.0, $P<0.001^*$	$F(3, 43285)=117.7$, $P<0.001^*$
	CPUE	$F(3, 43277)= 1389.5$, $P<0.001^*$	$(F(3, 43277)=$ 420.7, $P<0.001^*$	$F(3, 43277)= 45.0$, $P<0.001^*$

Mean number of individuals landed per trip was significantly different between boat types in the ZEPA (Kruskal-Wallis, Z= -26.7, P<0.001) and Tribugá (Kruskal-Wallis, Z= 79.7, P<0.001) with motorized boats landing more individuals than non-motorized boats (Table 4.4) (Figure 4.10A). Mangrove species showed a significantly higher number of individuals in landings than non-mangrove species in ZEPA (Kruskal-Wallis, Z=18.2, P<0.001) and Tribugá (Kruskal-Wallis, Z=73.3, P<0.001) (Table 4.4). Non-motorized boats in Tribugá landed a higher mean (± SE) number of individuals of mangrove associated species with 22.2 ± 1.7, compared to 8.3 ± 0.1 for non-mangrove species (Figure 4.10B).

Table 4.4. Mean and standard error for weight of fish and shellfish landed per trip, Catch per Unit Effort (CPUE), and number of individuals in landings per trip according to categories of mangrove dependence and boat type for Northern Chocó region between 2010 and 2013. Means (± SE) are provided separately for the Exclusive Zone of Artisanal Fishing (ZEPA) and the Gulf of Tribugá.

Dependent variable	Independent variable	Level	ZEPA	Tribugá
Weight landed per trip (Kg)	Mangrove dependence	Yes	44.0 ± 1.6	16.6 ± 0.2
		No	64.7 ± 1.0	19.8 ± 0.2
	Boat type	Non-motorized	17.5 ± 1.8	11.6 ± 0.1
		Motorized	61.2 ± 0.5	34.8 ± 0.2
CPUE (kg/fisher·day)	Mangrove dependence	Yes	3.5 ± 0.4	2.4 ± 0.08
		No	8.8 ± 0.3	4.9 ± 0.07
	Boat type	Non-motorized	8.7 ± 0.5	3.7 ± 0.06
		Motorized	6.5 ± 0.1	3.9 ± 0.09
Number of individuals landed per trip	Mangrove dependence	Yes	27.1 ± 0.8	23.3 ± 1.1
		No	24.9 ± 0.8	14.9 ± 1.1
	Boat type	Non-motorized	6.8 ± 1.1	15.2 ± 0.7
		Motorized	29.1 ± 0.4	30.6 ± 1.3

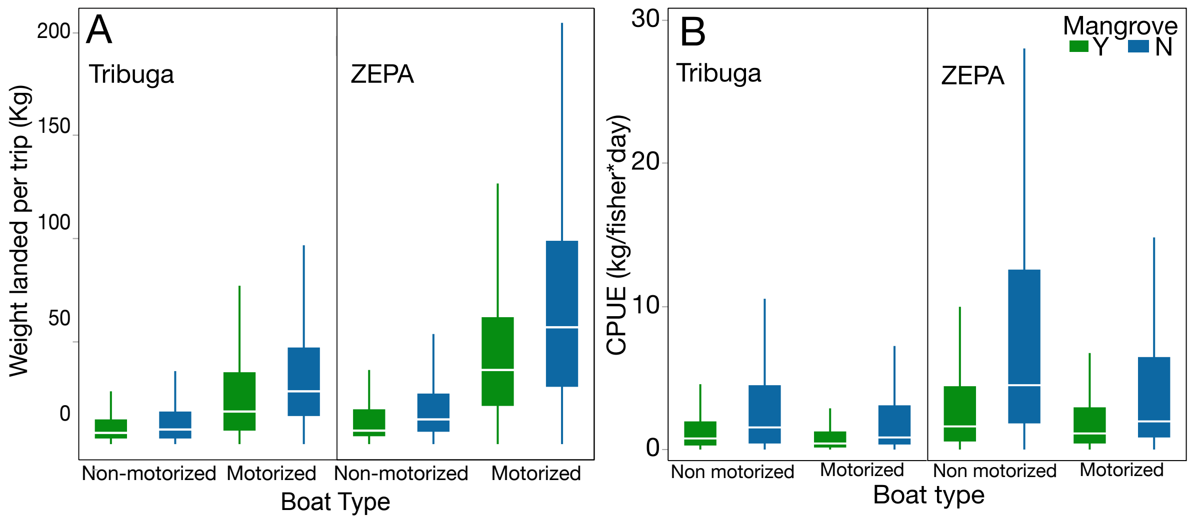

Figure 4.9. (A) Weight of fish and shellfish landed per trip and (B) Catch per Unit Effort (CPUE) of mangrove and non-mangrove associated species in different boat types across the Exclusive Zone of Artisanal Fishing (ZEPA) and the Gulf of Tribugá, between 2010 and 2013. In the boxplots, the line inside the box represents the median, and vertical lines indicate the range of the data inside the upper and lower quartiles.

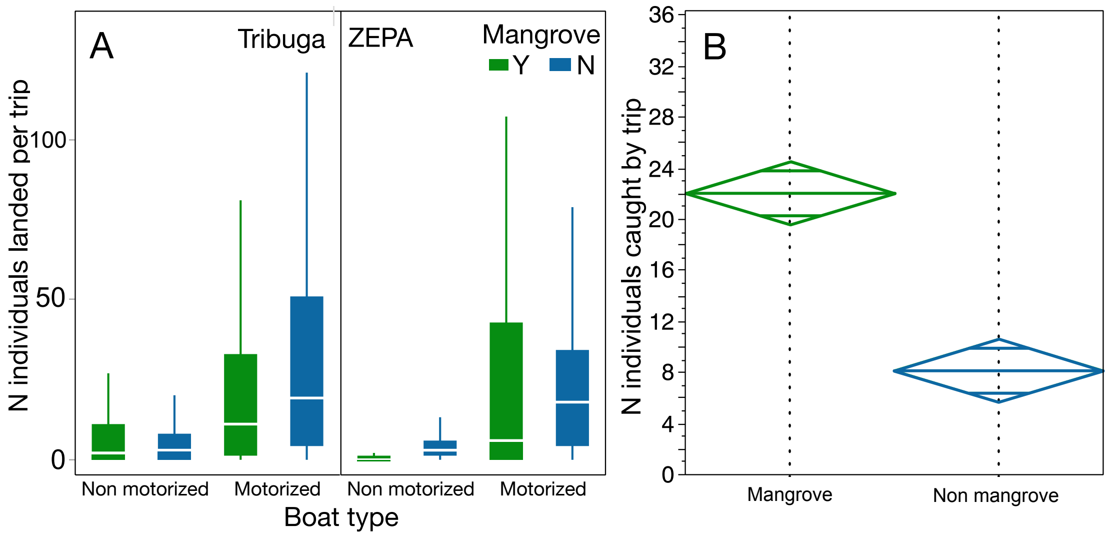

Figure 4.10. (A) Number of individuals landed per trip of mangrove and non-mangrove associated species in different boat types, across the Exclusive Zone of Artisanal Fishing (ZEPA) and the Gulf of Tribugá, between 2010 and 2013. In the boxplots, the line inside the box represents the median, and vertical lines indicate variability inside the upper and lower quartiles. (B) A closer view of the number of mangrove and non mangrove species captured by trip for non-motorized boats in Tribugá. Diamonds represent the mean and 0.95 confidence intervals.

Ark clam (*Anadara tuberculosa*) was the most abundant species landed by non-motorized boats with 39% of the total, followed by spotted rose snapper (*Lutjanus guttatus)* with 9.7%, green jack *(Caranx caballus)* with 9.1%, and flathead grey mullet (*Mugil cephalus*) with 8.6%. To remove the dominant influence of number of ark clams, I compared mean number of individuals landed between mangrove and non-mangrove species in non-motorized boats, excluding all records associated with ark clam. Mangrove species still showed significantly higher abundance than non-mangrove species (Kruskal-Wallis, $Z = 32.31$, $P<0.001$).

4.4. Discussion

Small-scale fisheries in Northern Chocó target a diverse array of inshore and offshore species using hand lines as the main gear. The ZEPA, where TURF management was established in 2008, showed significantly higher fisheries productivity than Tribugá, with CPUE increasing by 50% within the 3 years of sampling, despite evidence of gillnets still in use. Use of motorized boats was more prevalent in the ZEPA and were used to fish further offshore, contrasting with Tribugá, where non-motorized boats fishing closer to the shore were predominant. Non-mangrove species dominated landings by weight in the ZEPA and had higher CPUE than mangrove species. However, mangrove species were particularly important by number of landed individuals for small man-powered boats in Tribugá. Hence, it appears that in the ZEPA, management has driven fishing effort away from coastal habitats, and into pelagic zones, thereby increasing productivity and reducing pressure on mangrove resources. Findings suggest that the effects of area-based management in the ZEPA has generated an increase in CPUE, but at the same time, the effects of displacement of the industrial shrimp trawling fleet to Tribugá, cannot be ignored as a potential factor in driving a reduced CPUE in the artisanal fleet.

TURFs provide opportunities for improving and maintaining the welfare of small-scale fishing communities in developing countries, because they prevent the damaging consequences of open access, common pool resources, and allow an economically efficient use of these resources (Christy 1982). Therefore, in locations where there has been depletion of local marine resources, TURFs have been increasingly implemented (Cancino et al. 2007). In Japan, the TURF system encompasses most of the nation’s coastline and emerged in 1949 in response to overfishing and fisher conflicts (McIlwain 2013). Following a co-management scheme, resources are administered between the national government and fisheries cooperative associations by establishing catch, size and gear regulations and area closures, for a wide range of species including sardines and crabs (Wilen et al. 2012). Established more than 60 years ago, the system has continually enhanced the sustainability of fish stocks (McIlwain 2013). In Ecuador a TURF system was implemented in 1999 in mangrove fisheries, providing members of fisheries associations access privileges for mangrove cockles (Beitl 2012). After 15 years, Beitl (2016) found higher mean CPUE and larger shell size of mangrove cockles in TURF systems compared with open access sites.

Temporal trajectories indicate that CPUE increased in the ZEPA through to the end of the sampling period, whereas the opposite was true from the first year of sampling in Tribugá where there was no management (Figure 4.8). This suggests that the establishment of the ZEPA TURF in 2008 has led to improvements in fish stocks following past overfishing through i) exclusion of industrial fishing vessels and ii) prohibition of gillnets. A similar recovery pattern was observed in Chile, where TURFs were implemented in the sea snail (*Concholepas concholepas*) fishery following its collapse between 1989 - 1992 (San Martin et al. 2010). 10 years after the inception of TURFs, sea snail abundance and mean sizes of individual organisms increased, stabilizing catches and leading to increased public and private benefits (González et al. 2006; San Martin et al. 2010). In contrast, Tribugá fishers claimed that the establishment of the TURF in the ZEPA caused the displacement of the industrial shrimp trawling fleet to their waters (Díaz & Caro 2016). Fishing displacement is a known social consequence of the implementation of protected areas (Sen 2010; Cinner et al. 2014; Bennett et al. 2015; chollett et al. 2015), when their designation excludes fishers from access to their former fishing grounds (Charles 2009). Therefore, the greater pressure (fishing effort) displaced to the area left open to fishing can generate resource depletion, habitat degradation and/or socioeconomic consequences because of lower fishery profits (Hiddink et al. 2006; Gimpel et al. 2013). Hiddink et al. (2006) investigated the effects of area closures on benthic communities in the North Sea, and found that through displacement of effort some closures had negative effects on the overall biomass, production and species richness. This displacement of the industrial fleet's fishing activity may have caused the decline of CPUE of artisanal fisheries in Tribugá.

Former access to the ZEPA by shrimp trawlers likely resulted in significant habitat degradation and bycatch, as seen elsewhere (Freese et al. 1999; Hutchings & Reynolds 2004; Myers & Worm 2005; Hinz et al. 2009). Conflicts between industrial and artisanal fisheries in the same geographical space are a common issue in small-scale fisheries because of competition for resources (Bennett et al. 2001; Salas et al. 2007), and along with the use of gillnets, is one of the top problems faced by fishers in Colombia (Saavedra-Díaz et al. 2015; 2016). Declining trends of CPUE seen in the whitemouth croaker fishery in Uruguay were attributed to industrial fisheries activities overlapping spatially with small-scale fisheries (Horta & Defeo 2012). Given the indication that shrimp trawling may have intensified in Tribugá following exclusion from the ZEPA (Díaz & Caro 2016), research is needed to establish how this displaced fishing effort contributed to the decline in fish stocks in Tribugá. In the ZEPA, the formalisation of co-management structures and TURFs and the associated reduction in fishing pressure from industrial boats, appears to have enhanced success of gear-based management measures (Díaz & Caro 2016).

When fisheries move from open access conditions to rights based institutions, such as TURFs, associated economic revenues have been recorded up to five or ten times higher than those obtained in open access resource use (Wilen et al. 2012). In Japan and Chile for example, a significant amount of the rent generated under TURFs comes from market value improvements and better marketing opportunities via organization and cooperation (Cancino et al. 2007). Harvesting decisions in Chile's sea snail fishery moved away from unstructured deals with buyers to predictable sales, based not only on product quantity, but quality and market price (Cancino et al. 2007). In Colombia, a commercial connection between restaurants in Bogota and fisheries cooperatives was established soon after the ZEPA's formation, and has developed over recent years (Sáenz Pacheco 2014) with the support of many non governmental organizations and institutions (Cobos et al. 2016). Restaurants buy "responsibly-caught fish" directly from cooperatives in the ZEPA at a much more profitable price for fishers (Sáenz Pacheco 2014). This shorter market chain creates an incentive to diversify the fishery, targeting higher value species, with larger, more powerful boats (Cobos et al. 2016). In Northern Chocó, offshore sites harbour large pelagic species such as tunas, where productivity peaks in certain seasons according to specific fisheries (e.g. sardines) (Pereira Velásquez 1993; Zapata et al. 2007). The high mean weight landed of non-mangrove dependent species seen in the ZEPA is likely to be related to the dominance of motorized boats in this region (Table 4.2) as they operate further from the coast than non-motorized boats (Figure 4.6A). Tuna dominated landings by weight in the ZEPA, representing more than 50% of the total weight landed during the study period (Figure 4.3).

Tribugá's fishers have expressed their concerns over the decline of their catches and the increased sightings of trawlers (JM Díaz, *pers com,* Marviva Science Director). Following conflict with industrial fishers, stakeholders in the Gulf of Tribugá established the "Regional District of Integrated Management Golfo de Tribugá-Cabo Corrientes" in 2014 (Díaz & Caro 2016), introducing a spatial and temporal control over the activity of trawlers. The exclusion of the industrial shrimp trawler fleet from Tribugá may bring about a stock recovery, but the implications of a ban on gillnets for fisher communities with limited livelihood options and adaptive capacity are unknown, and likely to be variable (Tilley & López-Angarita 2016). Furthermore, the success of the ZEPA is also the result of financial investment to push fishing effort offshore and the development of fair trade style market chains between cooperatives and restaurants. Further studies in Northern Chocó should focus on comparing the effects of both management approaches (i.e. complete exclusion of trawlers vs. regulation of their activity) in the productivity of small-scale fisheries.

Landings of non-motorized boats in Tribugá were dominated by mangrove dependent species in terms of number of individuals (Figure 4.10B), although their contribution by weight was low compared to non-mangrove species (Figure 4.9A). The limited range of these boats means they operate near the coast and inside mangroves in Tribugá, so landed individuals of mangrove dependent species are likely to be juveniles, given the nursery function of mangroves (Nagelkerken et al. 2002; Mumby et al. 2004; Saenger & Funge-Smith 2013). Given the declining trends of fish abundance in Tribugá, it may be beneficial to reduce fishing pressure around mangroves to avoid growth overfishing. This could potentially increase fisher incomes and overall productivity by harvesting closer to the bio-economic optimum (Pollock 1993; Diekert 2012). Since non-motorized boats represent 75% of the fleet in Tribugá, nearshore and mangrove fish populations are clearly an important resource, so fostering their recovery is necessary.

For the Gulf of California, Aburto-Oropeza (2008), found that 32% of commercial fishery species were mangrove dependent, while this figure was 80% for commercial and recreational species captured in a study in Florida by Hamilton and Snedaker (1984). In my study around half of species landed were classified as mangrove dependent, with species diversity in landings closely related to gear diversity (Figure 4.5B). Gear diversity and target species diversity has been shown to be positively linked to the resilience of communities to environmental and management changes as it helps to reduce sensitivity to environmental impacts and management changes (Cinner et al. 2009; 2012; López Angarita et al. 2014; Tilley & López-Angarita 2016). Hand line was the dominant gear type in the ZEPA and Tribugá, but also it was the gear associated with the highest species diversity (Figure 4.5A). Hence, despite the tendency towards this single gear type potentially limiting resilience, the diverse array of species landed by hand line may compensate for this.

The higher catch rates demonstrated for ZEPA may also be due to an underlying geographical bias, as the Gulf of Tribugá has more estuaries compared to the ZEPA (Prahl et al. 1990). Unravelling the effects of management from other factors, such as sampling effort and environmental variability, is challenging given the lack of replication in this case study, where only two regions have been compared, and only one has management in place. The large difference in sampling effort between the regions may also affect comparisons, with more trips sampled in Tribugá. However, the dataset used is unique in Colombia in terms of length of sampling and detail of collected information. Furthermore, in the new management area established in Tribugá this dataset will prove invaluable as a baseline against which to track management success.

The TURF in the ZEPA did not eliminate industrial fishing pressure, but rather displaced it to affect neighbouring regions. For Tribugá, a local community initiative and the support of government and private organizations, have allowed the protection of artisanal fishers' livelihoods, but in other regions where fishing displacement occurs this might not be the case. Following management actions, it is necessary to contemplate the spatial and temporal redistribution of effort and use this information to predict potential impacts on livelihoods and natural resources (Hiddink et al. 2006). This planning, should prioritize small-scale fisheries over industrial fisheries given that the artisanal sector not only generates less impact with lower catches and higher selectivity, but also provides social benefits to a wider sector of the community through job creation and contributes more to local economies than the industrial sector (Belhabib et al. 2017). Leadership, social cohesion and co-management have been shown to improve well-being of small-scale fishing communities (Cinner et al. 2012) and promote successful fisheries (Gutiérrez et al. 2011). In the isolated fishing communities from Northern Chocó, where government resources are limited, the ZEPA has empowered local fishers through a system of fishing rights, management responsibilities, and rewards. The ZEPA not only represents a successful TURF, but also demonstrates the power of organization and cooperation of local community members and leaders. In the ZEPA, the TURF system reduced fishing pressure on coastal stocks, such as mangrove fisheries, by incentivising fishers to target offshore resources. As such, diverting fishing effort to higher productivity species offshore, a TURF system can effectively enhance mangrove fisheries protection for poor coastal fishers who rely on them for subsistence livelihoods. Finally, further socio-economic research needs to accompany fishery assessments to understand the effects of management on food security and poverty alleviation, which are areas of particular concern for the region studied.

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Chapter 5. Mangroves in the Eastern Tropical Pacific: deforestation trends, proximate drivers of land use and land cover change, and protected area effectiveness

Abstract

Mangroves are one of the most productive ecosystems in the world, sustaining millions of livelihoods in coastal areas. However, their area of occurrence has been greatly reduced over the last century. In Latin America most mangrove loss occurred between the 1960s and 1990s. In this study, I examine the conservation status of mangroves on the Pacific shorelines of Colombia, Panamá and Costa Rica, and calculate deforestation rates between 2000 and 2012, inside and outside protected areas. Proximate drivers of land use and land cover change adjacent to mangroves are identified via a cumulative model. Across all countries, agriculture was the land use type most often found adjacent to mangroves. Results show that only 564 ha of mangrove were lost from a total of 314,494 ha for the three countries combined, representing an average loss rate of only 0.02% per year. Comparing all the study countries, Colombia lost the least amount of mangroves with an annual rate of 0.01%, whereas Panamá lost the largest area with an annual rate of 0.02%. Costa Rica lost the highest percentage of total area, with 0.32% of their mangroves deforested at an annual rate of 0.03%. 75% of the total mangrove loss occurred in locations outside protected areas, with only 138 ha cleared from inside protected areas. Current conservation policies for mangrove protection in the study countries are effective, and set a positive example for regions where mangroves are in decline.

5.1. Introduction

It is estimated that by 2050 global crop production must double to meet demand from of a rising global population (Tilman et al. 2011). Improvements in crop yields, rather than a vast increase in acreage, is viewed as the best way to achieve this, however, regardless of progress in this respect, it is predicted that crop yields will still fall short of meeting projected global demand (Ray et al. 2013). Despite calls to not increase the world's area of cultivated land, a global pattern exists of increased agricultural field size driven by government incentives, demand for biofuels, and technology (White & Roy 2015). Worldwide rates of urban land expansion are higher than, or equal to, urban population growth rates (Seto et al. 2011). It is therefore expected that Land Use and Land Cover Change (LULCC) will increase as global population grows and developing countries become more affluent.

As land use intensifies to meet the high demand for commodities, the effects of arable land and urban land expansion on natural environments may have significant and potentially irreversible consequences in ecosystem function and integrity (Foley et al. 2005). These will be particularly intense in the tropics where species diversity and human reliance on natural environments are high, and LULCC are associated with agricultural products for food, feed, and fuel (Gibbs et al. 2010; Blanco et al. 2012). Here land conversion that removes primary forest has been shown to greatly reduce species diversity (Gibson et al. 2011), yet more than half the new agricultural land created between 1980 and 2000 was via deforestation (Gibbs et al. 2010). This conversion of natural habitat undermines ecosystem services from mangroves for food production, climate regulation, and freshwater resources, and thereby depreciates human welfare (Foley et al. 2005).

Mangrove forests are restricted to the interface between land and sea in tropical and subtropical latitudes. They are highly productive and provide a vast array of ecosystem services to people (Hogarth 2007) such as provision of nursery grounds for commercially important species (Nagelkerken et al. 2008) and highly efficient carbon sequestration (Donato et al. 2011). Moreover, mangroves diversify and sustain livelihoods for millions of people from the poorest sectors of society (UNEP 2014). Despite these widely appreciated values, mangroves are rapidly declining in different regions (Valiela et al. 2001; Alongi 2008; Richards & Friess 2015). Hence, there is a vital need to understand what drives deforestation and the ecological and social consequences of this.

Estimates of global mangrove loss vary across regions and with methods used (Alongi 2002; Giri et al. 2011; López Angarita et al. 2016). The development of optical remote sensing technology has allowed for better estimations of mangrove coverage, and for the exploration of LULCC dynamics (Manson et al. 2001; Dahdouh-Guebas et al.

2004). However, mangrove mapping can be challenging, as it is difficult to differentiate them from tropical rainforest in satellite images, and because some humid tropical regions have consistent cloud cover which affects clarity of satellite images (Gibbs et al. 2010; Heumann 2011). Recently, accurate estimates of mangrove deforestation rates have become possible thanks to the development of new radar technology that is sensitive to the spatial structure of forests and is not affected by clouds (Lucas et al. 2007; Simard et al. 2008; White & Roy 2015). However to date there is little study on the proximate drivers of LULCC in mangrove forests or replacement land uses (Tilman et al. 2011; Richards & Friess 2015).

In the Eastern Tropical Pacific (ETP), mangrove cover has followed global trends of decline, with its greatest loss occurring between the 1960s and 1990s (Valiela et al. 2001; López Angarita et al. 2016). Since then, here and worldwide, stronger calls for protection have occurred, coupled with increased understanding of mangrove importance as nursery habitat and for coastal protection (Valiela et al. 2001; Duke et al. 2007; Gibbs et al. 2010). Consequently, countries in the ETP have strengthened their conservation policies for mangroves, via creation of protected areas, and laws regulating mangrove use (Lacerda et al. 1993; ANAMARAP 2013; López Angarita et al. 2016). In general, LULCC in mangrove areas of the ETP has primarily been driven by development of shrimp aquaculture (Páez-Osuna 2001; Suman 2002; Gibson et al. 2011), although conversion to agriculture and coastal development for tourism and private property have also played major roles (Chapter 3) (Kaufmann 2012; ANAMARAP 2013).

In this study, I calculate rates of mangrove deforestation inside and outside protected areas on the Pacific coasts of Panamá, Colombia, and Costa Rica, between 2000 and 2012, to determine the effectiveness of mangrove conservation policies. To identify the proximate drivers of threat to mangroves in the countries studied, I map anthropogenic activities of LULCC in mangroves and perform analyses by country to compare trends within the region.

5.2. Methods

5.2.1 Study region

In this study I focus on ETP mangroves of Costa Rica, Panamá, and Colombia (Figure 5.1). This region, spanning the continental shelf and oceanic islands of Southern Baja California to northern Perú (Briggs 1974), supports a range of rich fisheries and exhibits many endemic species (Zapata & Ross Robertson 2006; Fiedler & Talley 2006; Hogarth 2007).

According to national datasets for the study sites, Costa Rica contains 23 Protected Areas for mangroves, Panamá has 17 and Colombia has 6 (Appendix 5.1). From these, a higher proportion of protected areas are designated under no-take management in Costa Rica and Panamá, compared to Colombia where limited-take areas prevail (López Angarita et al. 2016). 58% of mangroves in Costa Rica within the ETP are inside protected areas, compared to 51% in Panamá and 28% in Colombia (López Angarita et al. 2016).

5.2.2. Mangrove forest loss

To calculate rate of mangrove deforestation I used the Global Forest Change dataset created by Hansen et al. (2013), which provides an index of annual deforestation between 2000 and 2012 per pixel (30x30m). Given this dataset does not discriminate between forest types, I identified mangrove areas by overlaying the political limits of the studied countries with the global distribution of mangroves in 2000 provided by Giri et al. (2011). As I detected a projection error in the Giri et al. (2011) global distribution of mangroves dataset, where mangroves did not properly align with a section of the coastline and political boundary of Colombia, I corrected for this manually by fitting mangrove area polygons to the coastline, using as reference the most recent dataset of mangrove distribution for Colombia (IDEAM et al. 2007) and the satellite imagery of Google World Imagery (Esri, DigitalGlobe, GeoEye, Earthstar Geographics, CNES/Airbus DS, USDA, USGS, AEX, Getmapping, Aerogrid, IGN, IGP, swisstopo, and the GIS User Community, 2016). After corrections were made, I obtained a data layer of mangrove deforestation by year for the region of interest, based on Hansen et al. (2013). I then calculated the percentage of mangroves deforested in each individual country I studied. The rate of deforestation per year was obtained by dividing the percentage lost in the 11-year period covered by the deforestation dataset. All analyses were performed in ArcGIS 10.3.1.

Figure 5.1. Geographical extent of the study (red line) on the Pacific coasts of Costa Rica, Panamá, and Colombia (shaded green).

5.2.3. Proximate drivers of LULCC in mangrove areas

To map the distribution of proximate drivers of LULCC I used 10 different datasets of land cover across the three countries (Table 5.1). I grouped proximate drivers into three major classes: aquaculture, agriculture, and coastal development. Coastal development included presence of towns and infrastructure such as ports and agricultural processing plants. Infrastructure was not analysed as a separate class due to the few records associated with it. Datasets were visualized in ArcGIS and evaluated for correct land cover classification in areas adjacent to mangroves by cross-referencing with Google Earth imagery.

To quantify the spatial distribution of proximate drivers of LULCC adjacent to mangroves, I used an overlaid 1km^2 grid to divide mangroves of the region into sample units. Ground truthing was used to calibrate the interpretation of land use in Google Earth images. Ground truthing trips were made in 2013-2015 to all countries studied. I chose areas to ground truth based on them appearing to have significant presence of anthropogenic activities close to mangroves. Places visited were the Nicoya peninsula and the Terraba Sierpe wetland in Costa Rica; Aguadulce district and the Gulf of Montijo in Panamá; and the Gulf of Tribugá and Bahia Solano in Colombia. To obtain additional information about direct anthropogenic impacts on mangroves, I travelled by boat to locations inside the forests to visually determine mangrove state. Table 5.2 shows the percentage of 1km grid cells placed over mangroves of the studied region, where ground truthing occurred. In Costa Rica, I visited 13.6% of the cells in the study as a whole and these figures were 5.5% for Panamá and 0.2% for Colombia (Table 5.2).

Table 5.1. Details of the layers used to analyse proximate drivers of land use and land cover change adjacent to mangrove forest on the Pacific coastlines of Colombia, Costa Rica and Panamá.

Country	**Variable**	**Name of layer and source**
Colombia	Land use and land cover	National Cartographic database 2000 - 2009. National layer of land cover (CORINE land Cover) 2005-2009. Instituto Geográfico Agustin Codazzi
	Mangroves	Continental, coastal and marine ecosystems of Colombia. (IDEAM et al. 2007)
	Protected areas	Limites de áreas protegidas. Parques Nacionales Naturales de Colombia. 2014 -2015
Costa Rica	Land use and land cover	Atlas Nacional de Costa Rica (2008). Global land cover - GlobeLand30 (Chen et al. 2015). www.globallandcover.com
	Mangroves	Inventario Forestal de Costa Rica 2005.
	Protected areas	Sistema Nacional de Áreas de Conservación (SINAC) 2013
Panamá	Land use and land cover	Land use Panamá 2008
	Mangroves	Forest cover inventory 2000
	Protected areas	Sistema Nacional de Áreas Protegidas (SINAP)

Ground-truthing was used to improve image interpretation accuracy not only in the places visited, but also in other parts of the coast where land classification in the dataset did not match what was observed during field visits, or, for where the dataset was not of high enough resolution. When errors were found in the land use classification of the datasets, polygons were re-classified using Google Earth images calibrated with ground truthing. For example, in some cases, maps for aquaculture were not present in the datasets, or were present but had an incorrect land use classification assigned to them. In these cases, I used ground truthing to manually create maps of aquaculture by delimiting aquaculture ponds using Google Earth imagery calibrated by ground truthing.

Table 5.2. Total number of 1km² grid cells of mangroves in the 3 study countries, and the number of cells ground truthed per country.

Country	Total number of cells	Number of ground truthed cells
Costa Rica	1225	166
Panamá	4066	244
Colombia	4521	11
Total	9812	401

To display the spatial patterns of proximate drivers of LULCC adjacent to mangroves, I developed a cumulative model. Whereby I overlaid the 1km² grid with land use datasets, and extracted land cover information for each 1km² cell. From this I calculated the proportion of cells where aquaculture, agriculture, and coastal development were present throughout the region. This way a given cell could have one, two or three proximate drivers of LULCC present at the same time, while others had none.

For the cumulative model, each 1km² cell was given a score based on the presence (1) or absence (0) of each proximate driver of LULCC. Scores were summed per cell to return a possible value between 0 and 3. From this the proportion of cells belonging to different values of cumulative scores were calculated. Weighted cumulative models based on expert knowledge (Halpern et al. 2007; 2008; 2009) or stakeholders (Chapter 3), provide results based on the context of the area of interest. However, in this study because of the complexity of measuring the cascading effects that proximate drivers of LULCC have in mangroves on a regional scale, the same weighting was applied to all drivers. For visualisation purposes, cells were given a colour scale according to their total score, and were mapped to represent the pattern of proximate drivers adjacent to mangroves. All the analyses were performed using ArcGIS 10.3.1.

5.2.4. Protected areas

I compared the extent of mangrove deforestation inside and outside protected areas between 2000 and 2012, to estimate the effectiveness of protection legislation. To achieve this, I mapped the boundaries of protected areas present on the Pacific coast of the countries studied using government datasets (Table 5.1). Figures of mangrove deforestation within protected areas were estimated using the global forest change dataset (Hansen et al. 2013). Protected areas established after the year 2000 were analysed separately to accurately assess how deforestation had occurred inside and outside during the study period. Finally, I compared the distribution of proximate drivers of LULCC inside and outside protected areas by estimating the proportion of cells in each driver's class. Cumulative score of mangroves was also compared inside and outside protected areas. Analysis and calculations were executed in ArcGIS 10.3.1.

5.3. Results

5.3.1 Mangrove Forest Loss

Table 5.3 shows figures for mangrove deforestation in the study sites between 2000 and 2012. Over the study period, 6268 patches of mangroves were lost from the three countries combined. This equated to 564 hectares or 0.18% of the total mangrove area lost in 11 years (Table 5.3). In Costa Rica by 2012, 0.32% of mangroves present in 2000 had been deforested, with figures of 0.21% for Panamá and 0.11% for Colombia.

Table 5.3. Figures for mangrove deforestation between 2000 and 2012 on the Pacific coasts of Costa Rica, Panamá and Colombia.

Country	Total mangrove area in the Pacific coast (Ha)	Mangrove area deforested (Ha)	% of total area deforested	Annual deforestation rate (% of total area)
Costa Rica	37266.5	120.4	0.32	0.03

Panamá	135955.8	287.7	0.21	0.02
Colombia	141271.6	156.0	0.11	0.01
Total	314493.8	564.1	0.18	0.02

For the three countries studied combined, the average annual deforestation rate was 0.02%. Temporal trends of mangrove deforestation showed that deforestation peaked in Panamá and Costa Rica in 2008. An increasing trend of forest loss was observed in Colombia, whereas in Costa Rica deforestation has decreased with time (Figure 5.2).

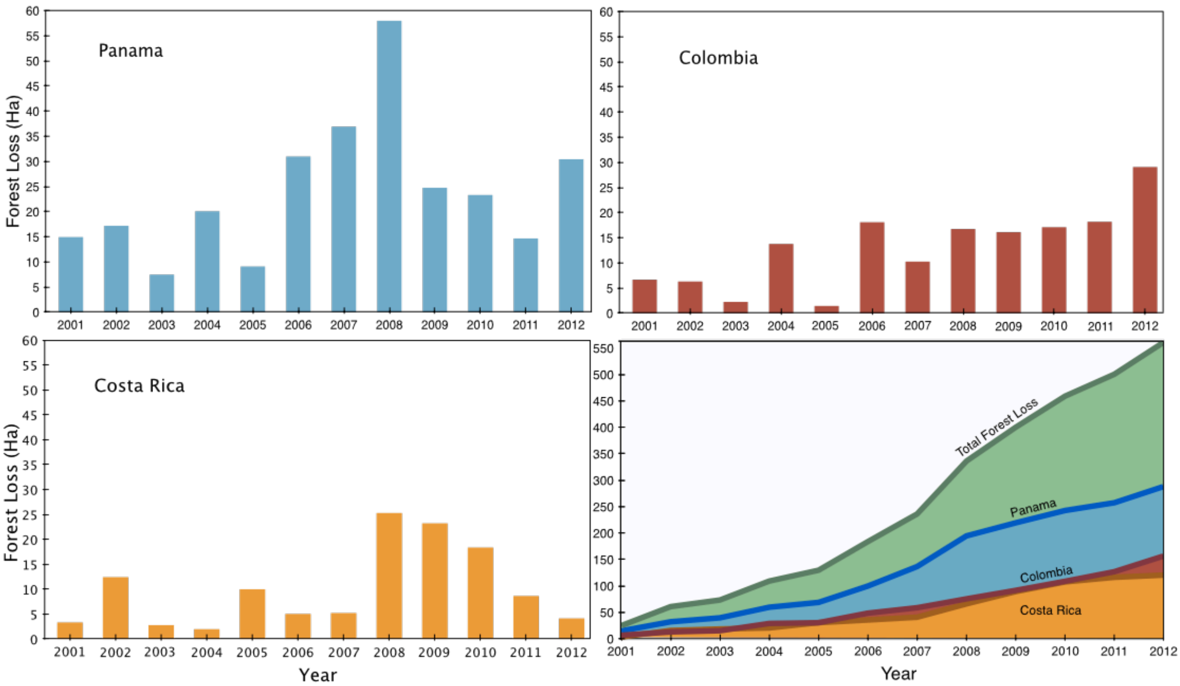

Figure 5.2. Temporal trends in the deforestation of mangroves between 2000 and 2012 on the Pacific coasts of Panamá, Colombia, and Costa Rica. The lower right panel shows cumulative forest loss for all countries.

5.3.2. Proximate drivers of LULCC in mangrove areas

According to the cumulative model of proximate drivers of LULCC in mangroves (Figure 5.3), around 60% of cells in the three countries combined had no adjacent proximate drivers of LULCC, whereas in 40% of cells, one or more drivers were present. In Colombia, 73% of cells with mangroves had no adjacent proximate drivers, whereas in ~26% of cells, one or more drivers occurred (Table 5.4). In this country the most common proximate driver of LULCC was agriculture, present in 26% of mangroves within the grid, with coastal development barely present and aquaculture not occurring (Figure 5.4). In Panamá, the cumulative model showed that proximate drivers were present in more than half of cells with mangroves (57%) (Table 5.4). Agriculture was the most dominant proximate driver, present in 30% of cells, followed by coastal development in 19%. 6% of cells were adjacent to aquaculture ponds (Figure 5.4). In Costa Rica, 60% of mangrove cells had no adjacent proximate drivers of LULCC, with one or more present in the remaining 40%. Agriculture was the most common land use, present in 28% of cells with mangroves. 9% of mangrove cells were adjacent to aquaculture ponds and another 9% adjacent to areas of coastal development (Figure 5.4).

Table 5.4. Detailed results for a cumulative model of proximate drivers of land use and land cover change adjacent to mangrove forests on the Pacific coasts of Costa Rica, Panamá and Colombia. Figures are calculated as percentages of 1km² grid cells placed over the mangroves.

Variable	**Colombia**	**Costa Rica**	**Panamá**
Cumulative model result	**% of cells**	**% of cells**	**% of cells**
No drivers present	73.4	60.6	42.6
One driver present	25.7	32.2	40.9
Two drivers present	0.9	6.8	15.7
Three drivers present	0	0.4	0.8
Proximate driver of LULCC			
Aquaculture	0	9.8	6.1
Agriculture	25.9	27.8	29.4
Coastal Development	1.6	9.4	19.4

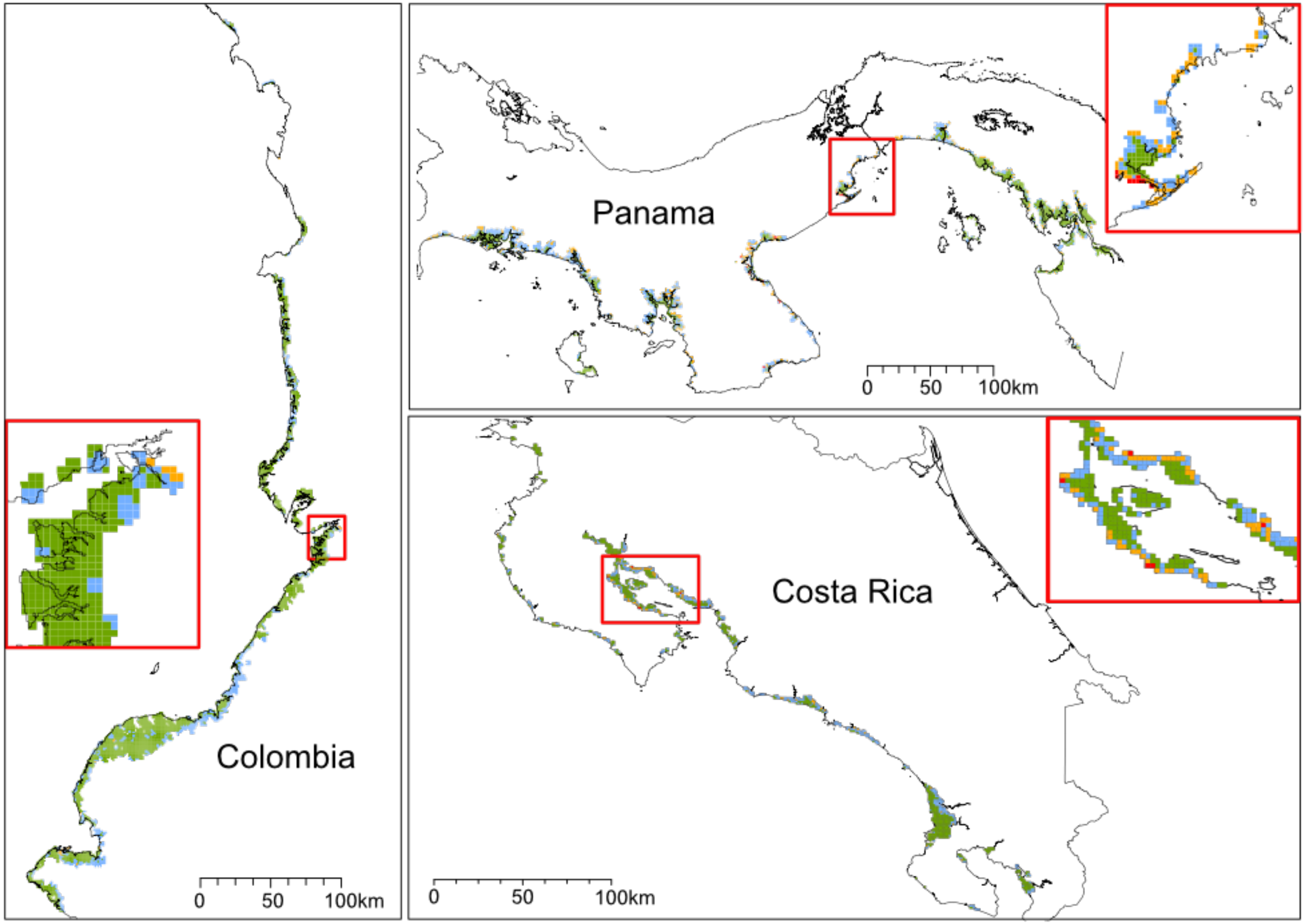
Colombia
0
50
100km
Panama
0
50
100km
Costa Rica
0
50
100km

Figure 5.3. Map of a cumulative model of proximate drivers of land use and land cover change in mangroves on the Pacific coasts of Costa Rica, Panamá and Colombia. Colour grid represents 1km² cells for which the analysis was performed. Green cells represent mangroves without adjacent proximate drivers of land use and land cover change, blue cells represent mangroves adjacent to one proximate driver, yellow cells are mangroves adjacent to two drivers, and red cells to mangroves adjacent to three drivers. For visualization purposes only, red insets provide a magnified view of the selected area.

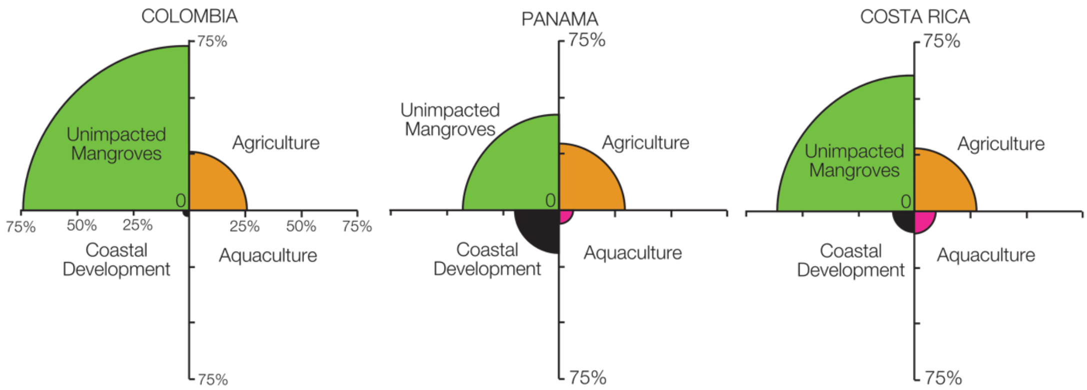

Figure 5.4. Distribution of agriculture, aquaculture, and coastal development next to mangroves on the Pacific coasts of Costa Rica, Panamá and Colombia. Percentages are calculated from a 1km² grid placed over the mangroves of the Pacific coast of each country.

5.3.3. Protected areas

Of the 31 protected areas on the Pacific coast of Panamá, 17 contain mangroves; in Colombia 6 of 9 contain mangroves; and for Costa Rica, 23 of 53 (Appendix 5.1). While figures for deforestation inside and outside protected areas varied between the three countries, loss inside protected areas was lower than outside in all cases (Figure 5.5). Across all three countries 75% of deforestation occurred outside protected areas (Figure 5.5).

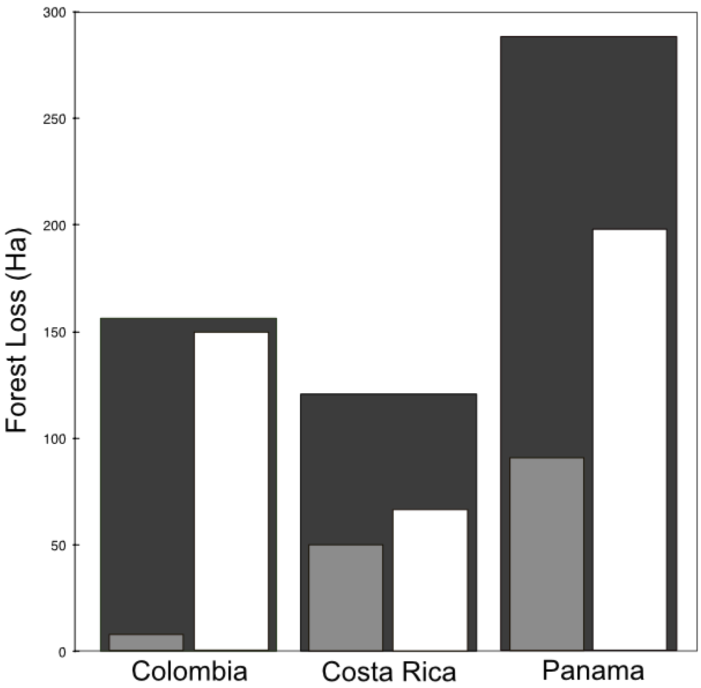

Figure 5.5. Total mangrove deforestation in hectares between 2000 and 2012 (black bars), highlighting deforestation inside (grey bars) and outside (white bars) protected areas on the Pacific coasts of Colombia, Costa Rica, and Panamá.

At the time of the research, 92% of cells with mangroves inside protected areas in Colombia had no adjacent proximate drivers of LULCC. In the remaining 8%, agriculture was present. In Panamá, agriculture was present in 50% of mangrove cells inside protected areas, whereas coastal development was present in 15%, and aquaculture in 6%. In Costa Rica, 68% of mangrove cells inside protected areas lacked proximate drivers of LULCC. Of the rest, 23% were surrounded by agriculture, and 8% by coastal development (Table 5.5).

Table 5.5. Detailed results for the cumulative impact model of activities adjacent to mangrove forests inside and outside protected areas of three countries of the Eastern Tropical Pacific. Figures are calculated as percentages of cells in a 1km² grid over area of mangrove occurrence.

Variable	**Colombia**		**Costa Rica**		**Panamá**	
Cumulative model result	**% of cells**		**% of cells**		**% of cells**	
	In	**Out**	**In**	**Out**	**In**	**Out**
No drivers	92	66.7	68.4	53.2	43.4	42.2
One driver present	7.7	32.1	26.8	37.3	41.9	40.3
Two drivers present	0.2	0.9	4.7	8.8	13.7	16.7
Three drivers present	0	0.2	0.2	0.6	0.9	0.7

Proximate driver of LULCC						
Aquaculture	0	0.4	4.7	14.7	6	6.1
Agriculture	8	32.2	23.6	31.7	50.4	18.6
Coastal Development	0.2	2	8.4	10.3	15.8	21.3

5.4. Discussion

Understanding recent trends of mangrove deforestation is important to evaluate the effectiveness of current conservation policies. However, as estimates of mangrove loss are not available for the ETP region, my study provides important information for managers in Colombia, Costa Rica and Panamá. The annual deforestation rate of 0.02% found for the three countries combined, supports the recent finding of a declining rate of mangrove deforestation of 0.16% per year at a global scale (Hamilton & Casey 2016). Moreover, the rates of mangrove loss found for each of the three countries in my study are confirmed by Hamilton & Cassey (2016), who used the same datasets to calculate global mangrove deforestation, but with a different methodological approximation.

Annual estimates of mangrove deforestation observed post-2000 are lower than estimates for previous years in the same locations. In the ETP, annual loss rates calculated prior to 2000 were higher than 1% for the three countries included in this study, because of figures for historic deforestation when most mangroves were lost (Valiela et al. 2001; López Angarita et al. 2016). Trends of mangrove deforestation found are consistent with historical trends, with Panamá displaying the largest losses and Costa Rica the lowest (López Angarita et al. 2016). Costa Rica however, showed the highest annual rate of loss in this study. This highlights the important point that loss rates can be misinterpreted if based upon the total area of cover lost, and not the rate at which mangroves are lost relative to the total area available. In 1998 Costa Rica became the first country in the ETP to declare no-take protection to all areas of mangrove (Valiela et al. 2001; RAMSAR 2001). Temporal patterns of mangrove deforestation illustrate that Costa Rica is the only one of my study sites to show a declining trend in mangrove area over recent years. By contrast Panamá and Colombia exhibit gradual increases in deforestation.

The rates of deforestation in the study countries are low compared to post-2000 deforestation in other regions and forest types (Valiela et al. 2001; Potapov et al. 2012; Nepstad et al. 2014; Richards & Friess 2015), which suggests that mangrove protection in the ETP is effective. By contrast, Richards & Friess (2015) estimated that between 2000 and 2010, mangroves in South East Asia were being lost at an average rate of 0.18% per year. For the same time period, in the Democratic Republic of the Congo, average annual gross forest loss was 0.23% across all forest types (Potapov et al. 2012). However, other studies have also reported significant declines in deforestation rates for other forests. In the Brazilian Amazon, forest loss declined 70% between 2005 – 2013, passing from a ten year average of 19,500 km^2 per year, to 5843 km^2 (Nepstad et al. 2014). National deforestation rates across all forest types decreased after the year 2000 in Costa Rica (FAO 2010), and Colombia (Cabrera et al. 2011). Perhaps due to its isolation, the Pacific coast region of Colombia shows the least amount of forest loss nationwide (Cabrera et al. 2011). Despite overall deforestation in Panamá having decreased compared to the 1990s, figures remain quite high for the region, with an annual rate of 0.41% between 2000 and 2008 (Mariscal 2012).

Data used in this study to quantify forest loss were derived from Landsat images with a resolution of 30x30m. It is possible that the spatial resolution used underestimates mangrove deforestation by not detecting losses at smaller scales. Deforestation is likely to happen at the interface between forests and other land use types (Etter et al. 2006), which makes it hard to detect in satellite images (Heumann 2011; Thompson et al. 2013). For example, in the Gulf of Montijo, Panamá, it was reported that the area of rice crops adjacent to mangroves has increased gradually (ANAM 2004) but the figures have not been quantified. During ground-truthing, no deforestation activities inside mangroves were observed, but many locals highlighted a slow expansion in the area of crops. There is underlying potential for underestimation of the integrity of mangroves, as satellite images at the scale used in this study do not allow for the identification of small-scale, or slow rate of habitat degradation at forest fringes. The recent development of high-resolution optical remote sensing sensors and techniques, allow for accurate mapping of diverse vegetation attributes such as species composition, dominance, above ground biomass, canopy closure, density, height and 3-D structure (Kamal et al. 2015; Zhu et al. 2015; Wang et al. 2016; Pau & Dee 2016), so may facilitate faster identification of small-scale deforestation.

Together, these variables can offer insight into mangrove health and status (Kamal et al. 2015; Giri 2016). Despite possible underestimates of deforestation in my study, my approach uses robust data of forest loss (Hansen et al. 2013) that has been applied to quantify forest cover change in many other regions worldwide (Potapov et al. 2012; Hansen et al. 2013; Richards & Friess 2015).

The cumulative model presented here serves as a tool to identify the most common proximate drivers of LULCC, and to detect areas for priority conservation action. Additionally, this model offers perspective on the threats affecting mangroves at national scales, determined by proximity of potentially damaging activities adjacent to mangroves. The proximate drivers identified here, are known sources of deforestation in Latin America (Geist & Lambin 2002; Achard 2002; Etter et al. 2006). My analysis found that agriculture is now consistently the most dominant proximate driver of LULCC adjacent to mangroves (Figure 5.4) outside and inside protected areas. Ground-truthing showed that rice, watermelon, melon, sugar cane, and oil palm are the main crops grown, and that cattle farming also occurs (Figure 5.6). The intensity and extent of agriculture adjacent to mangroves varied among the countries I examined, with small-scale agriculture prevalent on the Pacific coast of Colombia in contrast to more productive agro-economic regions found in Panamá and Costa Rica (Pinto & Yee 2011). In Panamá, rice and beef are the most commercially important agricultural commodities, and they are produced in rotation on the same land (Trejos et al. 2008). In Costa Rica, melon and oil palm have the highest yield per hectare, and these are grown in monocultures planted in high densities that receive large inputs of chemical pesticides and fertilizers (Bach 2007).

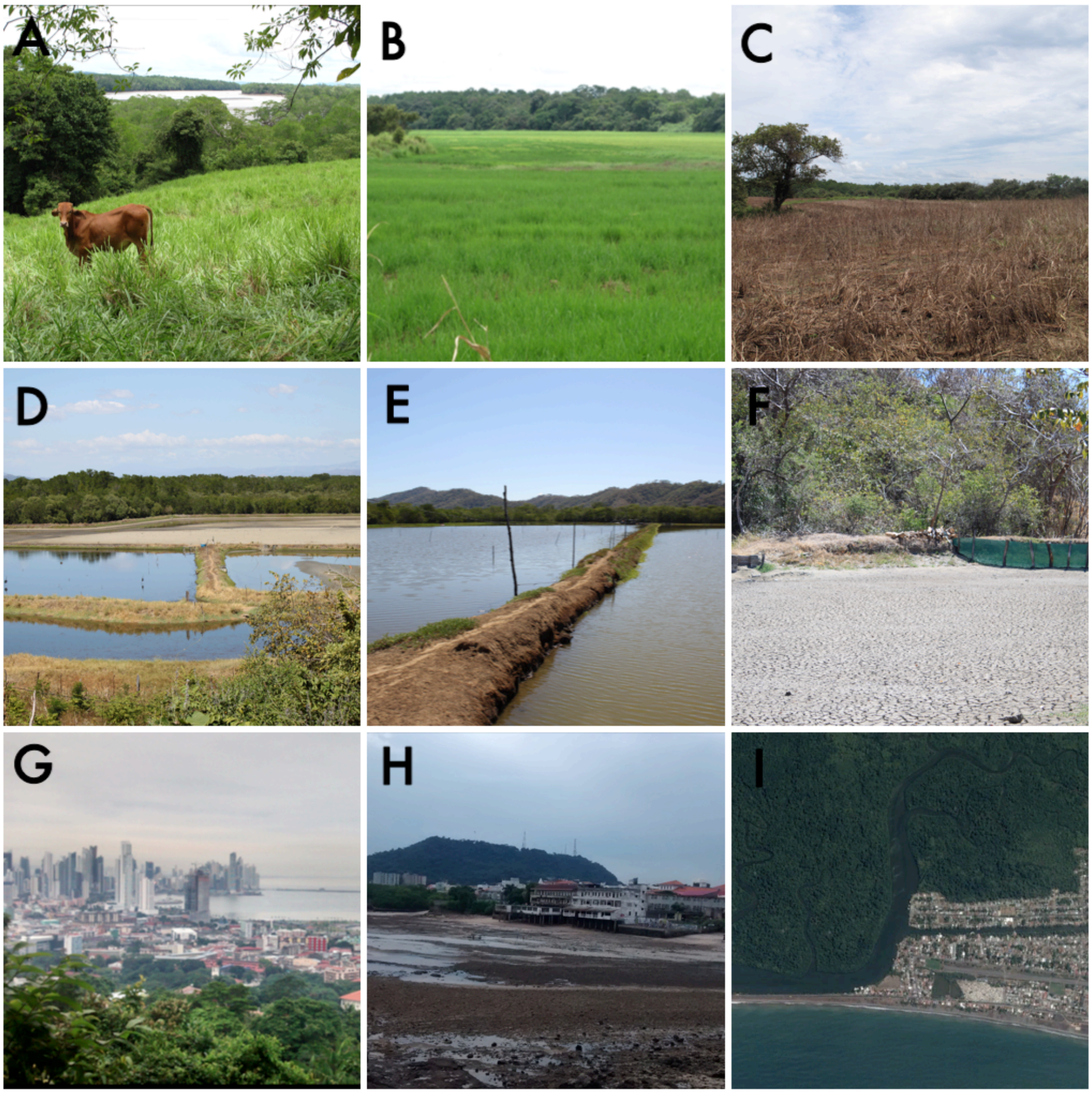

Figure 5.6. Photo panel showing proximate drivers of Land Use and Land Cover change adjacent to mangroves, seen during ground-truthing. (A) Fields used to graze cattle in the Gulf of Montijo (Panamá) are rotated with rice (B), which grows for 3 months until harvested (C). Shrimp farming takes place in ponds next to mangroves (D & E). Ponds are active for ~4 months then emptied and left to dry in the sun (F). Panamá City (G) is built on one of the biggest wetlands of the country and mangrove patches can still be seen around the city (H). (I) Aerial view of coastal development replacing mangrove forests in Puntarenas, Costa Rica © Google earth. Photos © Juliana Lopez-Angarita & Alex Tilley

Coastal development was a frequent proximate diver of LULCC in Panamá (Figure 5.6), whereas in Costa Rica it had the same prevalence alongside mangrove areas as aquaculture. Where mangroves are close to urban areas, they are commonly converted to areas of development (Benfield et al. 2005). Particularly for tourism and urban expansion this has been an important driver of mangrove clearance in Panamá (Kaufmann 2012). In Colombia, most of the Pacific coast population is scattered in small villages only accessible by boat (García 2010). This isolation from the rest of the country translates into good coverage of natural rainforest and mangroves, but simultaneously the lack of infrastructure and accessibility associated with this fosters poverty, and generates high dependence on natural resources (Leal 2000; Blanco et al. 2011).

Aquaculture, in particular extensive shrimp aquaculture, is widely claimed to be the most important driver of mangrove loss worldwide (Páez-Osuna 2001; Alongi 2002; Giri et al. 2008; 2011), but in the three countries studied, it was less important than agriculture. Between 1975 and 1985, Costa Rica lost ~350 hectares of mangroves to aquaculture. Shrimp production is semi-intensive and not of significant economic importance for the country, as pisciculture is more dominant in the contry (Wo Ching & Moreno 2001). In 1974, Panamá became the first country in the ETP to establish a commercial shrimp farming industry (Bolanos 2012), and the industry grew rapidly, such that 8100 ha were under production by 1998 (Suman 2002). However, in 1999 production declined sharply when the "white spot virus" invaded Central America, affecting both wild and cultured shrimp (Nunan et al. 2001). This caused widespread abandonment of ponds (Bolanos 2012), some of which have since been recolonized by mangroves (*pers. obs. 2014*), where soil conditions are suitable (Stevenson et al. 1999).

My analysis showed that most mangrove deforestation (75%) occurred outside protected areas. In Colombia, this equated to ~8 Ha, whereas the figures for Costa Rica and Panamá were 49 Ha and 81 Ha respectively (Figure 5.5). Other studies of tropical forests have also shown that the presence of protected areas significantly reduces deforestation inside them (Bruner 2001; Naughton-Treves et al. 2005; Andam et al. 2008; Gaveau et al. 2009; Miteva et al. 2015; Spracklen et al. 2015), and this has been shown specifically for mangrove forests (Miteva et al. 2015). Moreover, previous analyses found that protected areas in Costa Rica and Panamá were particularly effective in this regard (Andam et al. 2008; Spracklen et al. 2015). Andam et al. (2008) suggest that 10% of Costa Rican forests protected between 1960 and 1996 would have been deforested by 1997 in the absence of protection. Despite protected areas in this region being often undermanaged (López Angarita et al. 2014) and under increasing stress from human activities (Chape et al. 2005), my findings provide reassuring evidence that protection has had an overall positive effect in reducing mangrove deforestation. Despite the diverse management categories (e.g. national parks, forestry reserves, wildlife refuges) and approaches of protected areas included in this analysis (Appendix 5.1), all 3 study countries recognise in law that any activity intended to exploit or modify mangroves requires prior government evaluation and permission (García 2010; Salas et al. 2012; ANAMARAP 2013; López Angarita et al. 2016).

It has been shown that deforestation can be effectively reduced by the implementation of policy-driven measures such as: law enforcement, creation of protected areas, improved monitoring systems, and supply chain interventions for the triggers of forest clearance (Nepstad et al. 2014). In the ETP there has been an important movement towards mangrove protection, not only via top-down controls (López Angarita et al. 2016), but also through grass-root initiatives where conservation strategies generated by stakeholders have been highly effective for the sustainable management of natural resources (McClanahan et al. 2006; Moreno-Sánchez & Maldonado 2010; Gutiérrez et al. 2011; Cinner et al. 2012). In the ETP, communities have been reported to have increasing participation in the management of mangroves (Kaufmann 2012; Kothari et al. 2015; Vieira et al. 2016). I observed this during ground truthing via community driven reforestation programmes, schemes to protect mangroves against illegal deforestation, and in efforts to clean-up mangroves (Figure 5.7). Results presented here provide evidence that these initiatives combined with government input, are effective at reducing mangrove loss, and set a positive example for other regions where this ecosystem is being degraded.

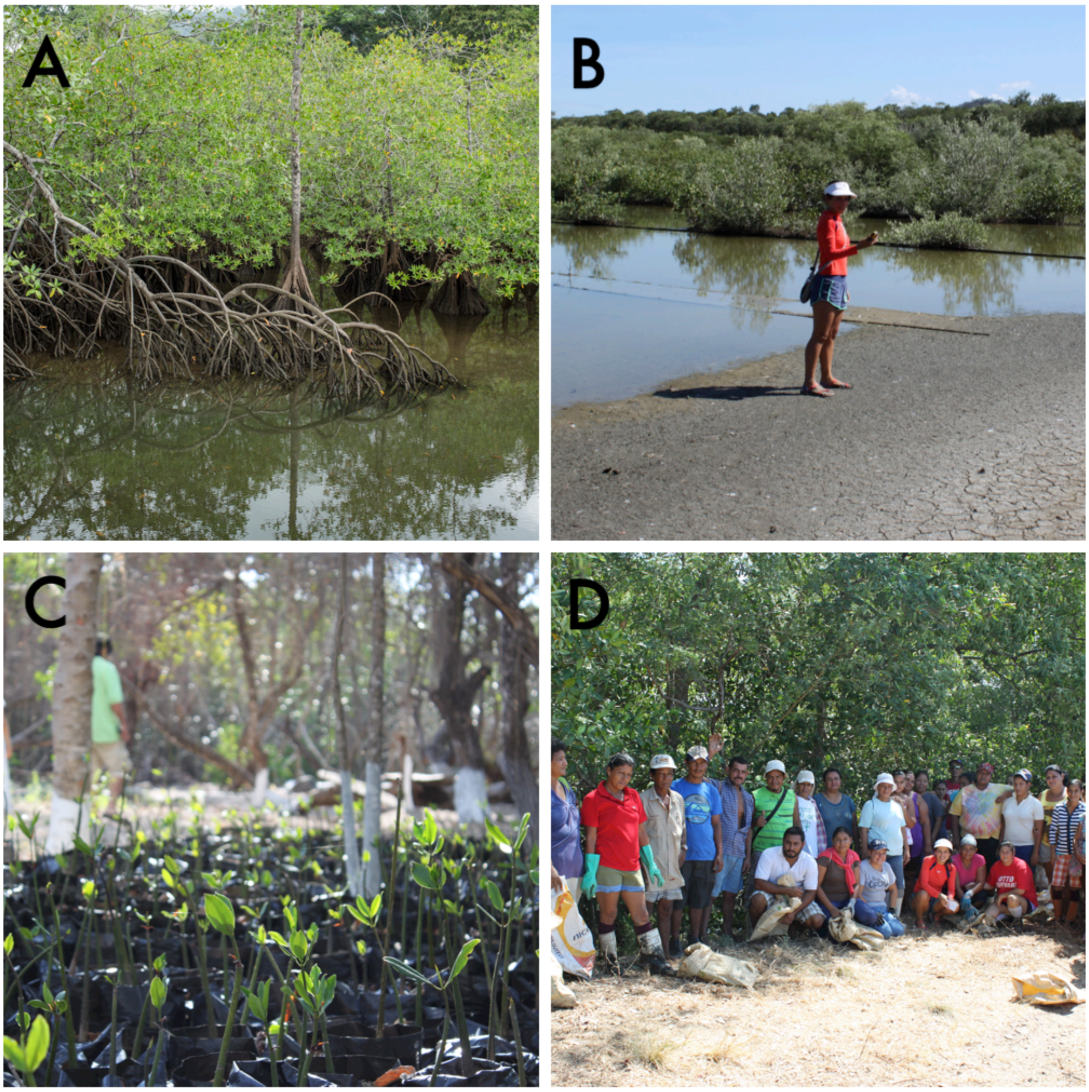

Figure 5.7. (A) A healthy mangrove forest in Utria National Park, Colombia. (B) Mangroves recolonising abandoned aquaculture ponds in Isla Chira, Costa Rica. (C) A red mangrove nursery as part of a community reforestation project. (D) The "Green Mangrove" community group during their daily activity of cleaning the mangroves of rubbish in Chomes, Costa Rica. Photos © Juliana Lopez-Angarita & Alex Tilley

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Chapter 6. General Discussion

6.1. Summary of thesis aims and results, and significance of findings

This thesis was developed to improve understanding of the conservation status of mangroves in the Eastern Tropical Pacific (ETP) region and the contribution of these systems to small-scale fisheries. My work tracked recent and historical decline trajectories of mangroves; identified the proximate drivers of mangrove deforestation and degradation; explored the contribution of mangrove species to artisanal fisheries; and evaluated the conservation effectiveness of extant/established protected areas at halting mangrove deforestation.

In Chapter 2, I explored the historical interactions between humans and mangrove forests in Colombia, Costa Rica, Panamá and Ecuador, and reviewed protected area coverage and existing policies for mangroves. My results highlight the importance of ETP mangroves for human societies since pre-Columbian times, with archaeozoological evidence showing that mangroves were exploited for many thousands of years. This is supported by studies around the globe showing that when present, mangroves provide important provisioning services and support artisanal livelihoods and income for millions of people by supplying forest products such as such as food, wood tannins, and honey (Dahdouh-Guebas et al. 2000; Walters et al. 2008; Baba et al. 2013). Mangrove area in the studied countries has declined almost 50% since first estimates from the 1950s and 1960s, and my study highlighted significant variability in area calculations over time, suggesting methods for this are still inconclusive. Recent research supports this, showing that binary pixel methods applied to calculate mangrove cover, overestimate mangrove area (Hamilton & Casey 2016). Using a high spatio-temporal resolution method to detect mangrove coverage, Hamilton and Casey (2016), found that their estimate of total mangrove area for the year 2000 at 83,495km^2, was 39% lower than the previous estimate of 54,360km^2 by Giri et al. (2011). In spite of these differences in mangrove area calculations, historical decline trends suggest the ETP has followed global tendencies despite their inherent variability. Research in other regions has shown that deforestation varies significantly among geographical scales, from nation to nation and even within nations (Hamilton 2013; Hamilton & Casey 2016). Drivers of deforestation are also highly variable. In the Indo-Pacific, conversion of mangrove to aquaculture has been shown to be the main cause of deforestation (Hamilton 2013; Richards & Friess 2015), whereas in the ETP, agriculture seems to be more prevalent (Chapter 2 and 5). My results show that in the ETP positive attitudes towards mangroves fuelled their inclusion in protected areas systems and conservation policies, from the 1990s onwards. However, protected area status does not always translate into effective protection. In Panamá, despite existing legislation, powerful economic interests drive urban expansion and the continued destruction of legally protected mangrove areas in Panamá Bay, one of the biggest wetlands of the country (Castellanos-Galindo et al. 2017). My findings for this chapter provide a baseline from which to understand present mangrove condition and call for a more central role for mangroves in national and regional conservation agendas.

In Chapter 3, I used the RAMSAR site of the Gulf of Montijo, Panamá, as a case study to explore how the condition of mangroves relates to their contribution to artisanal fisheries. By dividing the Gulf into 1km grid squares, I used Google Earth imagery and ground-truthing to classify land uses adjacent to mangroves throughout the Gulf. Land uses were then ranked according to level of impact on mangroves based on interviews with local fishers. Fisheries landings information for 15 communities showed that the percentage of mangrove dependent species found in landings was not significantly related to mangrove condition. Despite more than half of the species landed in the Gulf of Montijo being classified as mangrove dependent, the lack of relationship found illustrates the high degree of uncertainty from fisheries data (Mills et al. 2011; Saenger & Funge-Smith 2013) and the inherent difficulties of studying the link between mangrove condition and fisheries (Manson et al. 2005; Saenger & Funge-Smith 2013; Sheaves 2017). Interviewed fishers ranked agriculture, and rice production in particular, as the major threat to mangrove function and fisheries, given the lethal effect of fertilizers and chemical pesticides on fish populations. This is consistent with other countries such as Myanmar, where agriculture, particularly rice, has been shown to be an increasingly important driver of mangrove loss (Richards & Friess 2015). Despite the limitations and uncertainties of this case study, the findings of this chapter will be useful for managers of this and similar RAMSAR sites as it provides key information for developing a conservation plan for mangroves that benefits local communities. Furthermore, it provides a rapid characterisation framework to assess human impacts on wetlands in general.

In Chapter 4, I focused on the Northern Chocó region in Colombia, where an Exclusive Zone of Artisanal Fishing (ZEPA) was established in 2008. Management here follows the territorial use rights in fisheries (TURFs) system by excluding industrial shrimp trawling fishing fleets and prohibiting the use of nets. I examined the effect of this management on the catch composition of artisanal fisheries landings from 2010 to 2013 comparing the ZEPA area with the neighbouring unmanaged area of Tribugá. The ZEPA showed a recovery of fish stocks with increasing CPUE over time, whereas Tribugá remained stable after an initial decline potentially caused by absorption of displaced fishing pressure from industrial trawlers at the ZEPA. Furthermore, ZEPA landings composition showed much greater proportion of pelagic species. Fishing effort displacement following the implementation of area based management has also occurred in the North Sea (Hiddink et al. 2006), Australia (Sen 2010), and Kenya and Seychelles (Cinner et al. 2014). This chapter shows that TURFs can have significant positive effects on fish stock recovery but can also affect adjacent regions through fishing displacement.

Chapters 3 and 4 are an example of the challenge of studying the relationship between fisheries and mangroves. Many studies exploring this relationship have suggested that fish use mangroves in highly variable ways, and that differences in species composition or density of individuals over time, geographic locations, and multiple scales, is very common (Saenger & Funge-Smith 2013). Some argue that the generalized belief that a high percentage of fish use, or depend on, mangroves lacks the support of robust research (Sheaves 2017). Results from these chapters also illustrate the difficulties and uncertainties faced in fisheries studies. Scientific uncertainty about the state and productive capacity of the resource has been widely recognized as a pervasive feature of fisheries management (Charles 1998; Fulton et al. 2011; Pauly et al. 2013). Strong variations in recruitment, misreporting of catch, and sampling and processing mistakes in surveys, are frequent errors in fisheries management (Fulton et al. 2011). The use and interpretation of fisheries data is still debatable in the scientific community. Some dispute that fisheries catch data is a poor indicator of the abundance of fish because catch fluctuates in accordance with many factors, while others state that catch data is valuable as it is often the only resource available to inform management (Pauly et al. 2013). These issues are even more relevant for the highly dispersed and informal small-scale fisheries of developing countries, where data systems are generally under-resourced (Mills et al. 2011). In chapters 3 and 4, I analysed landings information of small-scale fisheries collected locally, and found limitations when analysing and interpreting the data. Whilst it is impossible to remove all uncertainty in fisheries research, the chapters' findings are a valuable insight for managers as they provide them with relevant information to manage resources, be it for the integration of mangroves with agriculture and fisheries, or for determining the consequences of area-based fisheries management. Existent efforts for monitoring artisanal fisheries in the ETP are useful and must continue to serve as an example for other areas in the region.

I extended the geographic range of my research again in Chapter 5 to investigate mangrove cover change trends between 2010-2012 in Colombia, Costa Rica and Panamá, and the effectiveness of protected areas at halting deforestation. I used open access sources of forest loss and mangrove coverage data to calculate temporal trends of deforestation inside and outside protected areas. Additionally, I analysed current land use in the areas and provided a regional map of accumulation of human activities adjacent to mangrove forests, to determine the potential impacts of land use on these ecosystems. My results indicate that mangroves have declined at an average rate of 0.02% per year, which is low compared to estimates from other regions, particularly South East Asia. Nevertheless, other studies exploring mangrove loss have also revealed a decrease in deforestation rates in the 21st century (Richards & Friess 2015; Hamilton & Casey 2016). In the countries I studied, a higher proportion of mangroves have been lost outside protected areas compared to inside, suggesting that protection policies are being effective. Protected areas have also been shown to be effective at reducing deforestation rates in other regions, with examples from Sumatra (Gaveau et al. 2009), Costa Rica (Andam et al. 2008), Indonesia (Miteva et al. 2015) and in many protected areas in tropical forests (Bruner 2001; Naughton-Treves et al. 2005; Spracklen et al. 2015). However, as discussed above, the presence of protected areas alone does not imply effective protection. Given that in recent years marine protected areas (MPA) have greatly expanded in number and scope, the challenge now lies in maintaining effective governance and protection success (Wells et al. 2016). Some have warned that the rush to establish MPAs without proper resources, impairs conservation by creating a false image of protection (Rife et al. 2012; Ray 2015). Rife et al. (2012) reviewed the MPA efficacy in the Gulf of California and found that most MPAs had not met conservation or sustainability goals. However, in this era of rapid environmental change the precautionary belief that MPAs can help reduce stressors and build resilience is essential (Ray 2015; Wells et al. 2016), particularly for vulnerable ecosystems such as mangroves. This chapter unravelled the temporal patterns of mangrove loss in the ETP, identified human activities that must be regulated to mitigate their impact on mangrove systems, and provided an analysis of the role of protected areas in mangrove conservation. These findings are useful for protected area managers in the ETP.

In a widely cited paper, Duke et al. (2007) raised concerns on the high rate of mangrove loss and estimated that the world could be without functional mangroves within 100 years. Fortunately, this scenario was based on extrapolated rates of mangrove deforestation from 1980s and 1990s and does not seem feasible now, as current research has shown that these trends appear not to have continued into the 21st century (Richards & Friess 2015; Hamilton & Casey 2016). Hamilton and Casey (2016) calculated that post-2000 mangrove deforestation decreased significantly, with some countries showing virtually zero mangrove loss. The difference between historical deforestation rates calculated in chapter 2 and modern rates estimated in chapter 5 provide evidence that this is also the case for the ETP. However, it is important to clarify that remotely sensed data at the scale of these studies, do not detect mangrove deforestation and degradation occurring at smaller scales, or the potential loss of mangrove function due to impaired ecosystem integrity. Mangrove deforestation has been shown to affect fish assemblages (Shervette et al. 2007; Shinnaka et al. 2007; Adite et al. 2013), alter the soil composition (Granek & Ruttenberg 2008; Hussain & Badola 2008), reduce carbon sequestration rates (Cebrian 2002; Irving et al. 2011), and decrease coastal protection against storms (Dahdouh-Guebas et al. 2005; Das & Vincent 2009; Yanagisawa et al. 2009). Particularly, loss of the nursery potential of mangroves will mainly affect people that depend on mangroves for their livelihoods and nutrition (MacKenzie 2001; Hutchison et al. 2014). Analysis of small-scale fisheries landings in chapters 3 and 4, highlighted the importance of mangrove associated species for small fishing communities in Panamá and Colombia. It is important to emphasize that despite findings in this thesis indicating that recent mangrove deforestation rates are lower than historical estimates, countries in the ETP need to strengthen their efforts of mangrove protection and sustainable management, given their importance for local livelihoods.

6.2. Future research

My work in the ETP revealed a need for baseline information on the conservation status of mangroves and fisheries in the region. Exploring the literature for historical estimates of mangrove area per country involved writing to government officers, who often stated that official figures did not exist. This is concerning because if governments do not have baseline data on forest cover, they cannot monitor change in natural forest area. This creates problems for designing policy to ameliorate human resource use with deforestation dynamics. While some estimates of mangrove cover do exist, they are not readily available and are spread across multiple sources. My research provides an up to date review and consolidation of these data, and it is recommended that local governments incorporate my figures in their repositories and official documentation and use them for conservation planning and policy making.

It is a recognized failing that governments often neglect to collect statistics for artisanal fisheries (Pauly & Zeller 2015; Schuhbauer & Sumaila 2016; Saavedra-Díaz et al. 2016) and this was true for the countries I studied, for which better information on landings is required. For example, in Panamá it has been shown that only 60% of fishery catches are reported (Harper et al. 2014). Given the similarities between the countries in the ETP, or even greater isolation of communities in Colombia, this is likely to be the case across much of the region, where underreported catches are the rule rather than the exception. Monitoring of small-scale fisheries is needed, given that they play an important role in nearshore resource decline (Díaz et al. 2011; Lindop et al. 2015), and many livelihoods depend on them (Smith et al. 2005). Monitoring artisanal fisheries will be the first step towards acknowledging the need to implement regulations and management with consideration of social-ecological contexts (López Angarita et al. 2014), both to prevent ecological degradation and highlight potential vulnerabilities in fisher communities (Tilley & López-Angarita 2016).

Further research to enhance understanding of key commercial fisheries species in the ETP is imperative, as the scarcity of this currently undermines our ability to provide quantitative and species-specific fishery management. Particularly relevant is the use of mangrove habitats during fish life history stages, as this can provide additional evidence to support the protection of mangroves for fishery purposes. While some evidence exists showing that mangroves can enhance fishery yields (Aburto-Oropeza et al. 2008; Serafy et al. 2015; Carrasquilla-Henao & Juanes 2016), the patterns of use of these by commercially important species needs to be better documented for the ETP as a whole (Saenger & Funge-Smith 2013). As part of this, fishers' ecological knowledge should be sought, as this will help provide good baseline information to guide new research (Castellanos-Galindo et al. 2011).

Finally, more work is needed to characterize the effect that agriculture, the major proximate driver of land use and land cover change identified here, has on mangrove ecosystems. It is essential to understand the underlying mechanisms by which agriculture can negatively affect mangroves. For example, in Panamá and Costa Rica fishers often report major die-offs of fish population in estuaries after pesticides and chemical fertilizers are used upstream (Trejos et al. 2007), but the effects of these in mangroves and fish are yet to be studied in the ETP. Tests of water quality near to agriculture fields should be a priority for governments in the ETP as part of their protection of coastal ecosystem integrity and for local livelihoods.

6.3. Conclusions

The ETP has the highest proportion of threatened mangrove species in the world (Polidoro et al. 2010) and also contains mangrove forests with some of the highest average biomass globally (Hutchison et al. 2014). The region is a biodiversity hotspot for many taxa (Myers et al. 2000) and contains pristine forests with low levels of human intervention (Global Forest Watch 2014). Ironically, financial resources for biological research in the ETP are limited by the prioritisation of other pressing development issues. Most of the existing information about mangroves in this region is contained within grey literature, not published in indexed or publically available journals and sources. More information is needed in the ETP to inform policy making to protect ecosystems such as mangroves.

My work provides baseline information for figures of mangrove area and temporal decline trends, and highlights the dependence of local fishing communities on mangrove associated species. This linkage is particularly precarious for poor fishers operating in areas without fishery management measures. My results show that most mangroves in the ETP were lost in the second half of the 20th century, whereas after the year 2000 mangrove deforestation decreased in response to mangrove protection policies. Given that the success of mangrove conservation depends on government capacity to integrate multi-institutional interests over mangroves, this thesis represents an important step towards promoting strategies for mangrove management that involve a better understanding of human-mangrove interactions.

6.4. References

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Appendices

Appendix 2.1. *Historical estimates of mangrove area for countries in the Eastern Tropical Pacific, years and references are provided.*

Year	Country	Area (Ha)	Source
1975	Costa Rica	39200	FAO, UNEP. 1978. Los bosques del país y su distribución por provincias. Costa Rica. Desarrollo Integral de los Recursos Forestales. Informe PNUD/FAO -COS/72/013. In: FAO. 2007b. Mangroves of North and Central America 1980-2005: Country Reports. Forestry Department. FAO. Forestry Department. FAO, Rome.
1979	Costa Rica	64452	Instituto Meteorológico Nacional. 1996. In: FAO. 2007b. Mangroves of North and Central America 1980-2005: Country Reports. Forestry Department. FAO. Forestry Department. FAO, Rome.
1983	Costa Rica	39000	Saenger, P., Hegerl E.J. and J.D.S., Davie. 1983. Global status of mangrove ecosystems. Commission on ecology Papers No.3. UICN. Gland, Suiza. 88 pp. In: FAO. 2007b. Mangroves of North and Central America 1980-2005: Country Reports. Forestry Department. FAO. Forestry Department. FAO, Rome.
1985	Costa Rica	40844	Herrera Wilbert. 1985. Tipo de clima de Costa Rica. In: FAO. 2007b. Mangroves of North and Central America 1980-2005: Country Reports. Forestry Department. FAO. Forestry Department. FAO, Rome.
1988	Costa Rica	41000	FAO. 1988. Proposed integrated forest management planning and utilization of mangrove resources in the Terraba-Sierpe reserve. Based on the work of Chong P.W. Informe técnico No. 2. Costa Rica Manejo integral de un area de manglar. TCP/COS/6652:FAODGF. 196 pp. In: FAO. 2007b. Mangroves of North and Central America 1980-2005: Country Reports. Forestry Department. FAO. Forestry Department. FAO, Rome.
1988	Costa Rica	37000	Spalding, M.D., Blasco, F. and Field, C.D., eds. 1997. World Mangrove Atlas. The International Society for Mangrove Ecosystems, Okinawa, Japón. 178 pp. In: FAO. 2007b. Mangroves of North and Central America 1980-2005: Country Reports. Forestry Department. FAO. Forestry Department. FAO, Rome.
1990	Costa Rica	30000	Furley, P.A. and Munro, D.M. 1993. The wetlands of Belize: ecology, environment and utilisation. Department of Geography, Edinburgh University, Edinburgo. 102 pp. In: FAO. 2007b. Mangroves of North and Central America 1980-2005: Country Reports. Forestry Department. FAO. Forestry Department. FAO, Rome.

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2005	Ecuador	150500	FAO. 2007c. Mangroves of South America 1980-2005: Country Reports. Pages 1–50. Working Paper 140. Forestry Department. FAO, Rome.

Appendix 2.2. *The main characteristics of the protected area systems of the countries in the ETP region.*

Country	*Land Area (km^2)*	*Marine Area (km^2)*	*Terrestrial Area protected (km^2)*	*Marine Area protected (km^2)*	*% of Area Protected*		*Number of PAs*	
					Terrestrial	*Marine*	*Terrestrial*	*Marine*
Costa Rica	*51,633*	*576,303*	*14,169*	*4,988*	*27*	*1*	*150*	*36*
Panamá	*75,893*	*332,373*	*15,614*	*43,292**	*21*	*13.5**	*69*	*25*
Colombia	*1,145,030*	*804,317*	*264,043*	*74,470*	*23*	*9*	*604*	*13*
Ecuador	*258,137*	*1,079,928*	*66,481*	*139,953*	*26*	*13*	*68*	*6*

*All data taken from www.protectedplanet.net. *Data provided by the Smithsonian Tropical Research Institute.*

Appendix 2.3. *Protected areas of the Tropical Eastern Pacific region (Ecuador, Colombia, Costa Rica and Panamá) that include mangrove forests within their limits. Data taken from* ***www.protectedplanet.net***

Country	Name of the Protected Area	Date of Creation	Management category*
Ecuador	Reserva Ecológica Manglares Cayapas Mataje	1995	No Take
Ecuador	Reserva Marina Galeras San Fransisco	2008	Mixed use
Ecuador	Refugio de Vida Silvestre Marino Costero Pacoche	2008	Mixed use
Ecuador	Parque Nacional Machalilla	1990	No Take
Ecuador	Reserva de Producción Faunística Marina Costera Puntilla de Santa Elena	2008	Limited take
Ecuador	Refugio de vida Silvestre Manglares El Morro	2007	Mixed use
Ecuador	Reserva de Producción Faunística Manglares el Salado	2002	Limited take
Ecuador	Reserva Ecologica Manglares Churute	1990	No Take
Ecuador	Refugio de vida Silvestre Manglares del río Muisne	2003	Mixed use
Ecuador	Refugio de vida Silvestre Manglares del río Esmeraldas	2008	Mixed use
Colombia	Parque Nacional Natural Utría	1987	Mixed use
Colombia	Parque Nacional Natural Uramba Bahía Málaga	2010	Limited take
Colombia	Sitio Ramsar del Río Baudó	2004	Limited take
Colombia	Parque Nacional Natural Sanquianga	1977	Mixed use
Costa Rica	Humedal Nacional Térraba Sierpe	1994	No Take
Costa Rica	Parque Nacional Piedras Blancas	1991	No Take
Costa Rica	Parque Nacional Corcovado	1975	No Take
Costa Rica	Parque Nacional Marino Ballena	1989	No Take
Costa Rica	Refugio Nacional de Fauna Silvestre Golfito	1985	Mixed use
Costa Rica	Reserva Forestal Golfo Dulce	1978	Mixed use
Costa Rica	Parque Nacional Manuel Antonio	1972	No Take
Costa Rica	Refugio Nacional de Vida Silvestre Isla San Lucas	2001	Mixed use

Costa Rica	Refugio Nacional de Vida Silvestre Playa Hermosa Punta Mala	1998	Mixed use
Costa Rica	Refugio de Vida Silvestre Cipancí	2001	Mixed use
Costa Rica	Parque Nacional Marino Las Baulas	1995	No Take
Costa Rica	Parque Nacional Palo Verde	1990	No Take
Costa Rica	Refugio de Vida Silvestre Camaronal	1994	Mixed use
Costa Rica	Refugio de Vida Silvestre Ostional	1983	Mixed use
Costa Rica	Refugio Nacional de Vida Silvestre Cipancí	2001	Mixed use
Costa Rica	Refugio Nacional de Vida Silvestre Caletas Arío	2006	Mixed use
Costa Rica	Refugio Nacional de Vida Silvestre Iguanita	1994	Mixed use
Costa Rica	Refugio Nacional de Vida Silvestre Mixto Conchal	2009	Mixed use
Costa Rica	Reserva Natural Absoluta Cabo Blanco	1963	Mixed use
Costa Rica	Parque Nacional Santa Rosa	1971	No Take
Costa Rica	Refugio de Vida Silvestre Junquillal	1995	Mixed use
Costa Rica	Humedal Estero de Punta Arenas y Manglares	2011	No Take
Panamá	Parque Nacional Darién	1980	No Take
Panamá	Humedal de Importancia Internacional Punta Patiño	1994	Mixed use
Panamá	Humedal de Importancia Internacional Bahía de Panamá	2003	Mixed use
Panamá	Parque Nacional Altos de Campaña	1966	No Take
Panamá	Parque Nacional Sarigua	1986	No Take
Panamá	Refugio de Vida Silvestre Pablo Barrio	2009	No Take
Panamá	Refugio de Vida Silvestre Isla Cañas	1994	No Take
Panamá	Refugio de Vida Silvestre La Cienaga del Mangle	1980	No Take
Panamá	Reserva Forestal La Tronosa	1977	Limited take
Panamá	Parque Nacional Cerro Hoya	1984	No Take
Panamá	Humedal de Importancia Internacional Golfo de Montijo	1994	Mixed use
Panamá	Refugio de Vida Silvestre Playa Boca Vieja	1994	No Take
Panamá	Parque Nacional Marino Golfo de Chiriquí	1994	No Take

Panamá	Refugio de Vida Silvestre Playa la Barqueta Agrícola	1994	No Take
Panamá	Parque Nacional Soberanía	1980	No Take

** No-take: No extractive activities allowed. Limited take: Some extractive activities allowed. Mixed use: A combination of limited-take and no-take.*

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Appendix 3.1. *GPS waypoints of the ground-truthing route along the Gulf of Montijo. Each waypoint has an associated description and some include a photograph for further reference. In certain cases the description contains more information than the one illustrated in the photo.*

No.	Description	Picture
1	Small enclosure for cows in a dairy farm.	
2	Cattle pasture.	
3	Cattle pasture.	
4	Puerto Mutis, is an important port that lies amongst the mangroves. I saw a relatively big artisanal fishing fleet of 25 boats on my visit. These logs were seen in the port, they are not mangrove wood.	
5	The Montijo landscape is a mosaic of human activities and natural forest. The hills used for cattle ranching are forested with scattered trees and lines of trees often divide property.	
6	A typical field of crops where a small area at the front is devoted to cassava and maize for personal consumption and to feed cattle, then behind it is a much bigger area of rice.	

7	An enclosure that appeared to be part of a dairy farm.	
8	High fences like this were not seen elsewhere. Inside this field cattle were grazing.	
9	This enclosure held the only goats I saw.	
10	Young rice has a very light green colour, and from afar looks like grass.	
11	Cows grazing in the front of a big house	
12	Tree plantation, probably balsa wood	
13	Rice crop	
14	Rice that has been just harvested. On the far side of the picture there are small oil palms which the farmer said the boss was trying to grow as an experiment.	

15	Rincon Redondo town. To the left going south there is a rice field. To the right there is a mountain.	
16	Sugar cane	
17	Quebrada Honda. This place can be seen clearly in Google earth, it is a big cattle transportation base.	
18	Cattle farm.	
19	Cattle farm, small lake nearby.	
20	River with rice crops on both sides. Picture taken from the bridge between “Las Almadanas” and “Las Hoacas”.	
21	Rice crops.	
22	Cattle ranch on the left going south and rice on right	
23	Cattle ranch and rice on right.	
24	A big rice farm. I spoke with the lady that lives here and guards the crop. She said that growing rice is the main activity in the GoM with some farmers also growing beans and cassava for their own consumption.	

25	A very large cowshed. On the left (going South), behind a pasture field.	
26	Sugar cane plantation. Some farms have turned to growing sugar cane for bio-fuel, since it is cheap to produce, requires less fertilizer and maintenance than rice, and takes one year to harvest.	
27	La Soledad is a little village with houses and small plantations of banana and cassava and football pitch	
28	Small corn plantation	
29	Recently harvested rice on the right, and young rice on the left (going south).	
30-31	Cattle farm very close to a mangrove forest which is just visible in the distance	
32	Rice on the right, and cattle on the left (going south).	
33	Rice crops	
34	Cattle farm	
35	Cattle farm on both sides of the road.	
36	Small lake for cows	

37	Recently harvested rice field. Once harvested they plant again, so before spreading the seeds (by hand) the field is burned so the seed reach the soil	
38	Cattle farm. On the left (mangrove side), moving south	
39	Rice. On the left (mangrove side), moving south	
40	Cattle. On the left (mangrove side), moving south	
41	Cattle. On the left (mangrove side), moving south	
42	Cattle. On the left (mangrove side), moving south	
43	Rice on left and right, then cattle. Moving south	
44	Timber plantation, for which it was not possible to identify the tree species.	
45	Wood plantation. This point is near the town called corosita, it is another wood plantation, had the sign "Forest Finance"	
46	Cattle farm	
47	Cattle farm. The landscape around this area is a very complex mosaic of use	
48	Road to La Playa	
49	Landscape to La Playa community.	
50	Cattle farm with cows just visible in the field.	
51	Hicaco town. End of road. South to the community and to the mangroves	

52	Cattle farms on both sides of road.	
53	Cattle farm	
54	Cattle farm with view of the GoM behind.	
55	Cattle farm infrastructure. Weeds at the start opposite the farm but then cattle both sides. Little farm and river along the way via Punta Calabazal.	
56	Burnt rice field on the way to Punta Calabazal.	
57	A massive rice field beyond which is mangrove forest from the tip of the Punta Calabazal peninsula	
58	On Punta Calabazal	
59	A mangrove forest very close to a field of rice, where the mangrove goes around the river that crosses the peninsula, and is not logged.	
60	Forest quite dense. Between Farfan and Tigre community, there are local houses, timber plantation (balsa wood) and weed. Very hilly landscape.	
61	Timber on right going north, cattle on left. Further ahead there is a little river	
62	Weed and corn beside a mangrove	

63	Cattle, lots of trees. Some corn behind.	
64	Cow pasture, forest behind with some other trees.	
65	Cattle both sides, mangroves in the distance	
66	Cattle field with mangroves and river estuary in the distance	
67	Cattle and rice being planted	
68	Forested area of bananas, palms, and houses. Mixed landscape with cattle beyond.	
69	Small and weedy field of corn with forest in the background on small scale, weeds in the field.	
70	Large cattle farm with mountains and estuary in the background.	
71	Cattle farm.	
72	Hibiscus. Mangrove forest	

73	River mouth near the town/village of Hibiscus.	
74	Cattle farm near the village of Lagartero	

Appendix 3.2. *Unstructured survey used to assess anthropogenic impacts on mangroves and importance of mangroves for fisheries in the Gulf of Montijo, Panamá. The interviews were filmed and designed to resemble a conversation guided by the interviewees' responses using the following questions as a guide.*

1. Describe the condition and health of the mangroves of the Gulf of Montijo? How does this compare with when you began fishing?

2. How clear and clean is the water around mangroves and in the Gulf? How does this compare with when you began fishing?

3. Do you think the amount of fish you catch relates to the condition of mangroves?

6. What is the most damaging impact on mangroves on your village?

7. Do you think the impact you mentioned affects the amount of fish you catch?

8. Are there certain species of fish you only catch in the mangroves? Are they the same species as when you began fishing?

9. Do you think mangroves have any important roles for you and your community?
If yes, what are they? Where or how did you learn about mangrove importance?

Appendix 3.3. *Species present in the Gulf of Montijo fisheries data set and classification of their reliance on mangroves.*

Common name	**Local Name**	**Species**	**Mangrove-dependent**	**Habitat**	**Reference**
Tuna	Atún	*Tunnus sp.*	No	Oceanic	1
Catfish	Bagre	*Bagre/Cathorops/Notarius/Occidentarius*	Yes	Coastal, brackish waters	1
Tripletail	Berrugate	*Lobotes pacificus*	Yes	Nursery area juveniles. Bays and estuaries	2
Long fin yellowtail	Bojalá	*Seriola rivoliana*	No	External area of reefs	1
Starry Grouper	Cabrilla	*Epinephelus labriformis*	No	Coastal waters in gulfs	1
Mangrove Grouper	Cherna	*Mycteroperca xenarcha*	Yes	Mangroves and estuaries.	1
Green jack	Cojinua	*Carax caballus*	No	Oceanic	3
Catfish	Cominate	*Arius platypogon, Arius kessleri, Ariopsis guatemalensis*	Yes	Coastal, frequent in soft bottoms	1
Catfish	Congo	*Cathorops fuerthii*	Yes	Sea, brackish and freshwater	1
Plink Brotula	Congrio (Merluza)	*Brotula clarkae*	No	Rocky shores and sandy bottoms	1
Weakfish/drum/croaker	Corvina	*Cynoscion/Corvula/Bairdiella/Larimus/Macrodon/Menticirrhus/Nebris/Ophioscion*	Yes	Coastal waters, lagoons, estuaries	1
Mahi mahi	Dorado	*Coryphaena hippurus*	No	Coastal and oceanic waters	1
Nurse shark	Gato	*Ginglymostoma cirratun*	No	Coastal demersal	1
Wahoo	Guajú	*Acanthocybium solandri*	No	Coastal and oceanic waters	1
Jack	Jurel	*Caranx*	No	Oceanic and coastal waters	1
Gey Mullet	Lisa	*Mugil curema*	Yes	Mangrove estuaries, brackish lagoons, coastal clear waters	4

Pacific Goliath Grouper	Mero	*Epinephelus quinquefasciatus*	Yes	Rocky, reefs, bays, estuaries and mangrove areas	1
Snapper	Pargos	-	Yes	Juveniles of this group usually found in mangroves and estuaries	1
Yellow snapper	Pargo amarillo	*Lutjanus argentiventris*	Yes	Juveniles in estuary	1
Pacific dog snapper	Pargo dienton	*Lutjanus novemfasciatus*	Yes	Rocky reefs. Juveniles may be encountered in estuaries with mangroves and at the mouths of rivers	1, 2
Colorado snapper	Pargo achiotillo	*Lutjanus colorado*	Yes	Coastal water hard bottom. Young often found inshore, sometimes in shallow coastal waters and estuaries	2
Spotted rose snapper	Pargo mancha	*Lutjanus guttatus*	Yes	Coastal water hard bottom. Juveniles inhabit estuaries and mouths of rivers	1, 2
Pacific red snapper	Pargo seda	*Lutjanus peru*	No	Sandy and hard bottoms	1
Mullet snapper	Pargo silguero	*Lutjanus aratus*	Yes	Juveniles in estuary	1
Rays	Raya	-	No	Muddy and rocky bottoms, and reefs	1
-	Revoltura	Mixed species of Weakfish, drum and croakers	Yes	Group common in estuaries and mangroves	4
Blackfin Snook	Robalo	*Centropomus medius*	Yes	Bays, rivers, estuaries	1
Doves longfin herring	Sardina	*Opisthopterus dovii*	No	Coastal water soft bottoms	1
Pacific Sierra	Sierra	*Scomberomorus sierra*	No	Coastal waters	1
Sharks	Tiburón	-	No	Oceanic	1
Whiteleg shrimp	Camarón blanco	*Litopenaeus stylirostris*	Yes	Muddy bottoms	5

Titi shrimp	Camarón titi	*Protrachypene precipua*	Yes	Muddy bottoms and river mouths	5
Cristal shrimp	Camarón rojo	*Farfantepenaeus brevirostris*	Yes	Sandy and muddy bottoms	5
Western white shrimp	Langostinos	*Litopenaeus occidentalis*	Yes	Muddy bottoms, juveniles in estuaries and adults are marine.	5
Lobster	Langosta	*Panulirus gracilis*	No	Rocky bottoms, between the intertidal zone and 22m deep	5
Cuata swimcrab	Jaibas	*Callinectes arcuatus*	Yes	Muddy bottoms in lagoon and estuarine systems	5
Inflated marsh clam	Almejas	*Polymesoda inflata*	Yes	Muddy bottoms and shallow waters influenced by fresh water	5
Panamá crescent Octopus	Pulpo	*Euaxoctopus panamensis*	No	Sandy and rocky bottoms	5
Cortez oyster	Ostión	*Crassostrea corteziensis*	Yes	Muddy bottoms of mangroves. Estuaries	5
Mussels	Mejillón	*Modiolus capax*	No	Soft or hard substrates, from low tide level until 50m deep	5
Black ark	Concha negra	*Anadara tuberculosa*	Yes	Abundant in mangroves, lives buried in mud	5
Polychaetes	Poliquetos	*Americonuphis reesei*	No	Inhabits muddy substrates	5
Conch	Cambute	*Stombus sp*	No	Inhabits a variety of soft bottoms	5

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Appendix 3.4. *Sample landing record for the community of Cebaco in 2010, showing how fisheries landing data was received. Numbers represent weight landed (lbs).*

Nombre comun	**Ene**	**Febr**	**Mar**	**Abr**	**May**	**Jun**	**Jul**	**Ago**	**Sep**	**Oct**	**Nov**
Atún											
Bagre											
Berrugate	9										
Bojalá											
Cherna											
Cojinua			68	159	129		387				
Congo	3,297	6,499	7,705	4,450	3,455	5,406	5,017	2,332	7,682	4,088	3,849
Congrio (Merluza)		532	244		545						903
Corvina	4,026	4,286	10,136	8,485	3,990	1,447	4,293	1,008	915	2,176	5,001

Dorado											
Gato											
Jurel									198		
Lisa	24		9								
Mero											
Pargos	136	369	383	402	59	60	531	0	349	295	2,230
Raya		1,311	57		351	4,924	3,896	1,731	12,147	3,995	2,799
Revoltura	7,018	8,534	15,715	15,898	16,791	5,018	9,801	4,436	1,680	3,111	6,089
Robalo							1,190			596	
Sierra	8,826	4,807	2,934	619	472	142	3,590	289	164	185	1,259
Tiburón		1,408	2,241	392	716	1,202	907	690	1,595	1,751	1,811
Camarón blanco	340			1,560	1,529	21	588				
Camarón titi				42	196	203		98		141	32
Langosta		124		9	9						
Jaibas										32	32
Almejas											
Ostión			400			600	800	600	600		
Pulpo								3			6
Mejillón										64	48

Lapas											
Cachicho											
Concha negra	600	600	1,200	930	500	600	360	400	800	856	800
Poliquetos											

Appendix 3.5. *Details of interviews performed in the Gulf of Montijo.*

No.	Village	Age	Affiliation
1	Puerto Mutis	67	Pescador Artesanal de Puerto Mutis
2	Puerto Mutis	64	Pescador Artesanal
3	Puerto Mutis	54	Pescador Artesanal
4	Puerto Mutis	58	Pescador Artesanal de Puerto Mutis
5	Santa Catalina	49	Pescador de Langosta
6	Santa Catalina	47	Pescador Artesanal
7	Santa Catalina	52	Pescador Artesanal
8	Guarumal	57	Asociacion de Pescadores Artesanales de Guarumal
9	Guarumalito	59	Asociacion de Pescadores Artesanales de Guarumalito
10	Guarumalito	37	Asociacion de Pescadores Artesanales de Guarumalito
11	Trinchera	49	Conchero y Trabaja en proyectos de reforestacion de Manglar
12	Hicaco	43	Pescador de la Asociacion de Pescadores de Hicaco
13	Hicaco	42	Administrador Asociacion de Pescadores de Hicaco
14	Hicaco	70	Pescador Artesanal
15	Lagartero	40	Asociacion de Pescadores Artesanales de Lagartero
16	Lagartero	42	Centro de Acopio de APODEAL Lagartero
17	El Pito	69	Asociacion de Pescadores Artesanales de El Pito

18	El Pito	57	Asociacion de Pescadores Artesanales de El Pito
19	La Playa	45	Asociacion de Pescadores Artesanales de La Playa
20	La Playa	52	Asociacion de Pescadores Artesanales de La Playa e Islas Leones
21	Palo Seco	57	Asociacion de Pescadores Artesanales de Palo Seco
22	Isla Cébaco	71	Pescador Artesanal
23	Isla Cébaco	56	Pescador Artesanal
24	Gobernadora	65	Pescador Artesanal de Gobernadora
25	Gobernadora	27	Pescador Artesanal de Gobernadora
26	Gobernadora	46	Pescador Artesanal de Gobernadora
27	Gobernadora	41	Pescador Artesanal

Appendix 3.6. *Cumulative impact scores values and category of impacts for the un-weighted and weighted models. Number of 1km^2 cells corresponding to each category are show.*

Un-weighted model			Weighted model		
Impact score value per cell	*Category of Impact*	*No. of cells*	*Impact score value per cell*	*Category of Impact*	*No. of cells*
0	Low	186	1	Low	186
1	Medium	317	2.4 4.1	Medium	303
2	High	128	4.4 5.5	High	19
3	Very High	1	5.8 8.9	Very High	124

Appendix 4.1. *Table of species found in fishery catches of the Northern Chocó region in Colombia, including scientific name, habitat, type of fisheries, common name, and classification of mangrove dependency (N: non mangrove dependent, Y: mangrove dependent). Information was obtained from the following websites: Fishbase (www.fishbase.org), The IUCN Red List of Threatened Species (www.iucnredlist.org), and Shorefishes of the Tropical Eastern Pacific (http://biogeodb.stri.si.edu/sftep/en/pages)*

Scientific Name	***Habitat***	***Fisheries***	***Common Name***	***Dependency***
Acanthocybium solandri	Oceanic waters	Commercial	Wahoo	N
Alopias pelagicus	Oceanic waters	Commercial	Pelagic thresher	N
Anadara tuberculosa	In mud sediments bellow the base of the roots of mangroves	Commercial	Cockle	Y
Anisotremus interruptus	Hiding in reefs	Commercial	Burrito grunt	N
Ariopsis seemanni	Common in coastal waters to depths of 20m, and enters estuaries	Commercial	Tete sea catfish	Y
Auxis thazard	Epipelagic in neritic and oceanic waters	highly commercial	Frigate tuna	N
Bagre panamensis	Inshore areas, muddy bottom	Commercial	Chilhuil sea catfish	Y
Bagre pinnimaculatus	Coastal waters	Commercial	Red sea catfish	Y
Balistes poliylepis	Rocky reefs	Commercial	Finescale triggerfish	N
Brotula clarkae	Juveniles in coral reefs, adults in deep waters	Minor commercial	Pacific bearded brotula	N
Brycon meeki	River basins	-		N
Calamus brachysomus	Reef associated, 3-18m	Commercial	Pacific porgy	N
Camaron titi	Muddy bottoms and river mouths		Protrachypene precipua	Y
Caranx caballus	Near the coast, Juveniles in estuarine waters	Commercial	Green jack	Y
Caranx caninus	Commonly found in coastal waters. Juveniles are often found in river estuaries	Commercial	Pacific crevalle jack	Y

Caranx otrynter	Oceanic waters	Commercial	Threadfin jack	N
Caranx sexfasciatus	Coastal waters & oceanic. Juveniles schools in estuaries and harbours	Commercial	Bigeye trevally	Y
Caranx speciosus	Deep lagoon and seaward reefs	Commercial	Golden trevally	N
Caranx vinctus	Oceanic waters	Commercial	Cocinero	N
Carcharhinus falciformis	Pelagic found near the edge of continental and insular shelves		Silky shark	N
Carcharhinus leucas	Continental and insular shelves	Highly commercial	Bull Shark)	N
Carcharhinus limbatus	River mouths and estuaries, muddy bays, mangrove swamps, lagoons, and coral reef	Commercial	Blacktip shark	Y
Cathorops steindachneri	Estuaries and river mouths	Minor commercial	Steindachner's sea catfish	Y
Caulolatilus affinis	Rocky and sandy bottoms	Commercial	Bighead tilefish	Y
Centropomus pectinatus	Coastal, entering estuaries and lagoons	Minor commercial	Tarpon snook	Y
Centropomus robalito	Mainly in estuaries	Commercial	Yellowfin snook	Y
Centropumus medius	Bays and estuaries	Commercial	Blackfin snook	Y
Cetengraulis mysticetus	Mud flats	Highly commercial	Pacific anchoveta	N
Chaenomugil proboscideus	Rocky littoral zones	Minor commercial	Snouted mullet	N
Chaetodipterus zonatus	Inshore areas, sandy & reef bottom	Commercial	Pacific spadefish	N
Chloroscombrus orqueta	Coastal marine and brackish waters, including lagoons with mangroves	Minor commercial	Pacific Bumper)	Y
Cirrhitus rivulatus	Bottom-living in shallow waters	Commercial	Giant hawkfish	N
Coryphaena hippurus	Near coast and open waters	Highly commercial	Common dolphinfish	N
Cynoponticus coniceps	Sandy and muddy substrate, as well as mangrove habitats.	Commercial	Red pike conger	Y
Cynoscion reticulatus	Coastal waters and estuaries with high salinities	Minor commercial	Striped corvina	Y
Decapterus macarellus	Oceanic waters	Commercial	Mackerel scad	N

Diapterus peruvianus	Juveniles in estuaries and mangroves	Commercial	Peruvian mojarra	Y
Elegatis bipinnulata	Coastal waters and oceanic waters	highly commercial	Rainbow runner	N
Elops affinis	Penetrate lagoon & estuaries	Minor commercial	Pacific ladyfish	Y
Epinephelus acanthistius	Sandy bottoms 46-90m	Minor commercial	Rooster hind	N
Epinephelus analogus	Reef associated	Minor commercial	spotted grouper	Y
Epinephelus analogus	Reef associated	Minor commercial	spotted grouper	N
Epinephelus cifuentesi	Rocky reefs usually in deeper waters	Highly commercial	Olive Grouper	N
Epinephelus labriformis	Shallow waters	Minor commercial	Starry grouper	N
Epinephelus quinquefasciatus	Juveniles in estuaries and mangroves	Minor commercial	Goliath grouper	Y
Eucinostomus currani	Juveniles in estuaries and mangroves	Minor commercial	Pacific flagfin mojarra	Y
Euthynnus affinnis	Close to the shoreline	Highly commercial	Kawakawa	N
Euthynnus lineatus	Coastal waters and offshore waters	Minor commercial	Black skipjack	N
Ginglymostoma cirratum	Continental and insular shelves, prop roots of red mangroves	Minor commercial	Nurse shark	Y
Haemulon maculicauda	Reef associated	Minor commercial	Spottail grunt	N
Haemulon steindachneri	Coastal rocky and coral reefs	Minor commercial	Chere-chere grunt	N
Halichoeres notospilus	Shallow water. Enter the mouths of estuaries on hard substrata.	-	Banded wrasse	Y

Hemanthias peruanus	Rocky reefs	Minor commercial	Splittail bass	N
Hemanthias signifer	Deep waters	Minor commercial	Damsel bass	N
Hoplopagrus guentherii	Hard bottoms in inshore reef	subsistence fishery	Mexican barred snapper	N
Hyporthodus niphobles	Rocky reefs and sandy bottoms		Star-studded grouper	N
Istiophorus platypterus	Oceanic waters	Commercial	Indo-Pacific sailfish	N
Katsuwonus pelamis	Offshore waters	Highly commercial	Skipjack tuna	N
Larimus argenteus	Coastal waters & lagoons	Minor commercial	common in markets	Y
Larimus effulgens	Coastal waters and lagoons	Minor commercial	Shining drum	Y
Lobotes pacificus	This coastal pelagic species occurs in bays and brackish estuaries	Commercial	Pacific tripletail	Y
Lutjanus aratus	Juveniles in estuaries and bays	subsistence fishery	Mullet snapper	Y
Lutjanus argentiventris	Inshore reef areas, brackish	Commercial	Yellow snapper	Y
Lutjanus colorado	Juveniles in estuaries	Commercial	Colorado snapper	Y
Lutjanus guttatus	Juveniles in estuaries	Commercial	Spotted rose snapper	Y
Lutjanus inermis	Coastal rocky and coral reefs	Commercial	Golden snapper	Y
Lutjanus jordani	Shallow mangrove-lined embayment	Commercial	Jordan's snapper	Y
Lutjanus novemfasciatus	Juveniles in estuaries and mangroves	Commercial	Pacific dog snapper	Y
Lutjanus peru	Hard bottoms in inshore reef	subsistence fishery	Pacific red snapper	N
Lutjanus viridis	Coastal rocky and coral reefs	subsistence fishery	Blue and gold snapper	N
Macrodon mordax	Coastal waters, bays, and estuaries	Commercial	Dogteeth weakfish	Y
Manta birostris	Shallow reefs, at the surface inshore and offshore.		Giant Manta Ray	N

Megalops atlanticus	Coastal waters, bays, estuaries, mangrove-lined lagoons, and rivers	Commercial	Tarpon	Y
Menticirrhus undulatus	Sandy shores and in bays	Commercial	California kingcroaker	N
Mugil cephalus	Coastal, entering estuaries and rivers	Highly commercial	Flathead grey mullet	Y
Mulloidichthys dentatus	Mud & rocky & sandy bottom. Found in estuaries and coastal lagoons	-	Mexican goatfish	Y
Mustelus henlei	Continental and insular shelves	highly commercial	Brown smooth-hound	N
Mustelus lunulatus	Continental and insular shelves	Commercial	Sicklefin smooth-hound	N
Mycteroperca xenarcha	Mangrove and estuaries	Minor commercial	Broomtail grouper	Y
Myripristis leiognathos	Rocky reefs	Minor commercial	Panamic soldierfish	N
Nematistius pectoralis	Shallow inshore areas	Minor commercial	Roosterfish	N
Oligoplites refulgens	Temporarily penetrate estuarine waters	Minor commercial	Shortjaw leatherjacket	Y
Ophioscion scierus	Shallow water. Occur in estuarine and mangrove areas	Minor commercial	Point-Tuza croaker	Y
Opisthonema libertate	Coastal waters and offshore waters	highly commercial	Pacific thread herring	N
Panulirus gracilis	Rocky and gravel-sand bottoms	Commercial	Green Spiny Lobster	N
Paranthias colonus	Reef associated	Subsistence fishery	Pacific creole-fish	N
Polydactylus approximans	Shallow water near the coast, on sand and mud bottom	Commercial	Blue bobo	Y
Polydactylus opercularis	Coastal waters and estuaries, on sand and mud bottom	Minor commercial	Yellow bobo	Y
Pontinus furcirhinus	Deep waters		Red scorpionfish	N
Pristigenys serrula	Deep waters		Popeye catalufa	N

Pseudopeneus vancolensis	Sandy bottoms	Commercial	Yellowfin goatfish	N
Sarda orientalis	Coastal waters	Minor commercial	Striped bonito	N
Sciadeops troschelii	Coastal waters/coast	Commercial	Chili sea catfish	Y
Scomberomorus sierra	Coastal waters and offshore waters	Commercial	Pacific sierra	N
Sectator ocyurus	Open water over deep reefs	Minor commercial	Bluestriped chub	N
Selar crumenophthalmus.	Oceanic waters	Highly commercial	Big eye scad	N
Selene brevoortii	Pelagic-demersal shallow coastal waters	Minor commercial	Hairfin lookdown	N
Selene brevoortii	Pelagic-demersal shallow coastal waters	Minor commercial	Hairfin lookdown	N
Seriola peruana	Pelagic-demersal coastal waters	Minor commercial	Fortune jack	N
Seriola rivoliana	Outer reef slopes and offshore banks	Commercial	Longfin yellowtail	N
Serranus huascarii	Demersal species found in depths of 80 to 200m	Not commercial	Flag serrano	N
Sphoeroides rosenblatti	Inshore waters to 4 m depth in brackish-water estuaries, amongst mangroves and river mouths		Oval Puffer	Y
Sphyraena ensis	Continental shelf	Commercial	Mexican barracuda	Y
Sphyrna lewini	Pups tend to stay in coastal zones, near the bottom, occurring at high concentrations during summer in estuaries and bays		Hammerheads	Y
Spondyliosoma cantharus	Seagrass & rocky & sandy bottom	Commercial	Black seabream	N
Stellifer mancorensis	Shallow water, sandy muddy bottom	-	Smooth stardrum	N
Strongylura scapularis	Coastal waters and lagoons with mangroves	Minor commercial	Shoulderspot needlefish	Y
Sufflamen verres	Rocky reefs	-	Orangeside triggerfish	N

Thunnus albacares	Pelagic	Highly commercial	Yellowfin tuna	N
Trachinotus paitensis	Coastal waters	Commercial	Paloma pompano	N
Triaenedon obesus	Sluggish inhabitant of lagoons and seaward reefs	Minor commercial	Whitetip reef shark	Y
Tylosurus crocodilus fodiator	Coastal waters	Commercial	Mexican needlefish	N
Umbrina xanti	Sandy bottoms	Commercial	Polla drum	N
Umbrina xanti	Sandy bottoms	Commercial	Polla drum	N
Xenichthys xanti	Shallow sandy bottoms	Commercial	Longfin salema	N

Appendix 4.2. *Fishing villages in the Exclusive Zone of artisanal Fishing (ZEPA) and the Gulf of Tribugá. Data is taken from the last official population census of Colombia performed in 2005 (www.dane.gov.co). The landing site of "Cabo Marzo" in the ZEPA was not included as the population is itinerant.*

Community	*Zone*	*Number of inhabitants*
Arusi	Tribugá	313
Coqui	Tribugá	150
Jovi	Tribugá	163
Jurubida	Tribugá	332
Nuqui	Tribugá	2759
Pangui	Tribugá	304
Partado	Tribugá	123
Termales	Tribugá	162
Tribugá	Tribugá	130
Bahia Solano	ZEPA	3077
Cupica	ZEPA	1050
Huina	ZEPA	152
Jurado	ZEPA	3881
Nabuga	ZEPA	271
El Valle	ZEPA	2653

Appendix 4.3*. Tukey-Kramer post-hoc comparisons of mean CPUE between all the gear types used in Northern Chocó. Levels not connected by the same letter were significantly different.*

Fishing gear	ZEPA	Tribugá
	Connected means	
Long line	A	A
Beach Seine	A B	A B
Spear gun	A B	B C
Gillnets	A	C
Hand line	B	C
Cast net	-	C D
Manual collection	A B	D

Appendix 5.1. *Protected areas of the Pacific coast of Colombia, Costa Rica, and Panamá included in this study.*

Country	Name	Management category	Year of creation	Area (km^2)
Colombia	Sanquianga	Natural national park	1977	866.85
	Utría	Natural national park	1987	653.67

	Uramba Bahía Málaga	Natural national park	2010	473.18
	Rio Anchicaya	National protective forestry reserve	1946	1451.52
	Territorio Colectivo	Regional district of integral management	2008	68.11
	Parque Natural Regional La Sierpe	Natural regional park	2008	252.97
Costa Rica	Caletas-Arío (mixto)	Wildlife refuge	2006	204.34
	Cipanci (estatal)	Wildlife refuge	2001	34.83
	Corcovado	National park	1975	444.9
	Estero Puntarenas y manglares	Wetland	2001	51.93
	Golfito (mixto)	Wildlife refuge	1985	28.19
	Golfo dulce	Forestry reserve	1978	599.9
	Iguanita (estatal)	Wildlife refuge	1994	1.13
	La Ensenada (mixto)	Wildlife refuge	1998	4.86
	Las Baulas de Guanacaste	National park	1991	273.25
	Manglar Térraba-Sierpe	Wetland	1994	261.83
	Manuel Antonio	National park	1972	1264.64
	Marino Ballena	National park	1992	53.6
	Ostional (estatal)	Wildlife refuge	1983	86.24
	Palo Verde	National park	1978	172.06
	Palustrino Corral de Piedra	Wetland	1994	24.27
	Pejeperro (mixto)	Wildlife refuge	2000	5.95
	Piedras Blancas	National park	1991	158.43
	Portalón (mixto)	Wildlife refuge	1996	2.24
	Playa Hermosa	Wildlife refuge	1998	27.89
	Río Oro (estatal)	Wildlife refuge	1999	17.17
	Santa Rosa	National park	1966	860.35
	Santuario Ecológico	Wildlife refuge	2003	3.31
	Tivives	Zone of protection	1986	24.74
Panamá	Bahía de Chame	Multiple use area	2007	89.00
	Bahia de Panamá	Wetland	2003	489.19
	Patiño	Wetland	1993	131.98
	Golfo de Montijo	Wetland	1990	864.76
	Coiba	National park	2005	2548.24
	Golfo de Chiriqui	National park	1994	212.21
	Sarigua	National park	1984	46.70
	Isla Cañas	Wildlife refuge	1980	242.85
	La Barqueta	Wildlife refuge	1994	67.04
	Cenegon del Mangle	Wildlife refuge	1980	8.43
	Playa Boca Vieja	Wildlife refuge	1994	35.79
	Pablo Barrio	Wildlife refuge	2009	150.32
	Chepigana	Forestry reserve	1960	363.79

	Canglon	Forestry reserve	1984	286.23
	Filo del Tallo	Hydrological reserve	1997	122.26
	Isla del Rey	Hydrological reserve	2006	98.22
	Archipiélago de Las Perlas	Special zone of marine coastal management	2007	1601.51