Ecological importance of nearshore habitats to sustain small-scale fisheries

by

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B.Sc., Jorge Tadeo University, 2008
M.Sc, Instituto Politécnico Nacional, 2011

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Abstract

In the marine realm, there has been considerable habitat degradation caused by multiple human disturbances that often act synergistically, strongly affecting fish and invertebrate populations and, consequently, one of the major stakeholders of these resources, fishers. However, the mechanisms underlying how marine habitats support fisheries remain understudied. In this dissertation I examined the importance of fish habitat at global, regional and local scales in two distinct systems (mangrove habitats in the tropics and rockfish habitats in inshore waters of Vancouver Island) combining a suite of different approaches. First, I explored the mangrove-fishery linkage relationship by conducting a global meta-analysis. I found strong evidence supporting the importance of mangrove area to enhance fisheries. This relationship, however, varied across countries, likely based on regional geomorphological settings and fishery management policies. Subsequently, I determined the use of mangrove and adjacent habitats by fish in a tropical lagoon system in the continental Caribbean (Colombia), systems often overlooked in the Caribbean when analyzing mangroves as fish habitat. I collected fish with gillnets at different distances from mangroves and at different sites within the same lagoon system. While fish used mangroves, fish abundance was not higher in these habitats compared to adjacent ones, as predicted. However, diversity tended to be higher in mangroves. Nevertheless, the major driver affecting abundance, diversity and biomass was salinity. That is, diversity and abundance decreased as salinity increased. Next, I used a Local Ecological Knowledge approach to explore the mangrove-fishery linkage relationship because fishers are seldom incorporated into such relationships. By conducting semi-structured interviews I found that fishers fish close to their village and to mangroves, that in addition to fishing they use mangroves for firewood and as construction material. Fishers also agreed that mangroves are important for their fishing activity, as these habitats are critical for fish and crustaceans caught in the system. Finally, I examined the importance of derived benthic parameters for rockfish abundance and distribution at large spatial scales (100s km) in inshore waters of Vancouver Island. I established that higher complexity better explains presence and higher abundance of rockfish. Furthermore, the results provided valuable information for fishery and spatial management and habitat conservation to help recover rockfish populations. All together, these findings highlight the urgency to preserve coastal marine habitats for both juvenile and adult marine organisms to sustain small-scale fisheries as a food source and for traditional purposes. While conserving habitats is a key component of a broader and more complex ecosystem approach that includes overfishing and other anthropogenic pressures, in the absence of a holistic approach the chances of success are minimal.

Table of Contents

List of Tables

List of Figures

Acknowledgments

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Dedication

To Jimena and Joaquin, the loves of my life

Introduction

1.1. Human impact in ecosystems and habitats

Globally, terrestrial and aquatic ecosystems are being deteriorated or lost at high rates driven by anthropogenic pressures primarily caused by deforestation (Bender et al. 1998) induced by the increase of agricultural areas (Vitousek et al. 1997; Kehoe et al. 2017). Climate change is a second factor affecting ecosystems (Hoegh-Guldberg and Bruno 2010) likely produced by a constant human population growth and high use of fossil fuels (Smith 2011; Isbell et al. 2017). As a result, biodiversity has decreased considerably (Butchart et al. 2010; Cardinale et al. 2012) to the extent that we are nearing a new mass extinction (Barnosky et al. 2011). This ubiquitous and cumulative trend is problematic because higher biodiversity enhances ecosystem function (Chapin et al. 2000; Cardinale et al. 2012; Lefcheck et al. 2015), and, as a consequence, the number and quality of ecosystem services has also decreased (Isbell et al. 2017). Furthermore, many of the species lost are foundation species that provide important habitats for other organisms. For example, trees are critical habitats for many bird species for nesting, feeding and resting purposes in terrestrial ecosystems (Fearer et al. 2007), while coral reefs, kelps and mangrove forests are important fish habitats in the marine environment (Ferrari et al. 2016). Although habitat loss and ecosystem deterioration in marine ecosystems is generally less than in terrestrial systems (McCauley et al. 2015), perhaps because impacts are more complicated to quantify and because until recently deep waters were inaccessible (Vitousek et al. 1997), marine habitats have also been heavily impacted for a number of reasons. The major threat to marine ecosystems, particularly near-shore systems, has been overexploitation, which has had profound consequences on ecosystem structure and functioning by altering natural food webs, community structure and shifts in the abundance and size structure of the harvested populations (Jackson et al. 2001; Venter et al. 2006; McCauley et al. 2015).

Habitat loss is the second most important threat to marine coastal systems and is caused by multiple stressors such as pollution (Halpern et al. 2008), urban development, port construction, water contamination, land modification and direct habitat destruction (Lotze et al. 2006; Venter et al. 2006). Habitat is the combination of resources and environmental conditions present that allows an organism to survive and reproduce (Hall et al. 1997). As a result, marine systems have also suffered a decrease in biodiversity (Worm et al. 2006). For example, mangrove coverage across the world has been diminished considerably to build extensive aquaculture farms and for agriculture (Valiela et al. 2001; Alongi 2002; Spalding et al. 2010; Duarte et al. 2013). Similarly, seagrass and kelp forests have declined due to high nutrient input and more frequent heat waves (Duarte et al. 2013). As a consequence, the lack of suitable habitat and habitat fragmentation has negative repercussions for marine organisms including fish and crustaceans because they are faced with limited areas to satisfy biological, physiological and ecological roles and thus, cause population declines.

Habitat loss exacerbates ecosystem deterioration because it alters the connectivity of organisms among habitats, a common ecological process in the marine realm, particularly based on ontogenetic shifts (Nagelkerken et al. 2015) whereby fish and crustaceans use one type of habitat as juveniles, known as nurseries, and then migrate to different adult habitats helping to maintain adult populations. Nursery habitats support large numbers of juvenile fish because they are usually structurally complex providing additional shelter, food and three-dimensional habitat (Ferrari et al. 2017). The most common nursery habitats in the oceans vary on a latitudinal gradient. For example, in the tropics, particularly in the Caribbean, mangroves are the most important nursery habitats for reef fish that migrate to coral reefs as adults (Igulu et al. 2014; Nagelkerken et al. 2017). Conversely, seagrass beds serve the same purpose in sub-tropical and temperate systems (McDevitt-Irwin et al. 2016). In the central coast of British Columbia, Rockfish of the genus *Sebastes* use seagrass and kelps as nursery habitats (Olson 2017). Thus, from a conservation standpoint preserving juvenile and adult habitats is equally important to allow such critical ontogenetic movements that may benefit fisheries. Likewise, understanding how organisms use such habitats can enhance how habitats are conserved. Unfortunately, how fish and other organisms use coastal habitats is not yet fully understood.

1.2. Human coastal population and fisheries

The rapidly growing human population (Smith 2011) aggregates in coastal areas across the world (Vitousek et al. 1997; Worm et al. 2006) exerting a higher pressure and impact on coastal ecosystems (Halpern et al. 2008). However, these coastal ecosystems provide multiple goods and services for human populations. For example, tropical estuaries, which are dominated by mangrove habitats, are among the most productive in terms of ecosystem services and fish production (Costanza et al. 1997; Blaber 2013).

Mangroves provide a number of wood and timber products for use in house construction and as firewood (Dahdouh-Guebas et al. 2006). Furthermore, mangroves provide coastal protection because they aggregate sediments and may be critical to mitigate climate change by dissipating wave energy in a sea level rise scenario (Duarte et al. 2013). In temperate regions, where mangroves are absent, seagrass meadows and kelp forests provide similar services though no wood products are obtained (Duarte et al. 2013; Efird and Konar 2014).

Though many ecosystem services are provided by nearshore ecosystems, perhaps the most pervasive is fishing, an occupation that has persisted for over thousands of years (Jackson et al. 2001). Initially fishing methods were rudimentary and effort was focused on coastal habitats. However, with the industrial revolution, several technological improvements occurred over several decades, which allowed fishers to make their activity much more efficient, but also facilitate overfishing otherwise healthy stocks (Jackson et al. 2001). As a result, many fish populations were overfished or fully exploited (Botsford et al. 1997). Overfishing is a current concern given the importance of fishery resources to supply food to the human population (Béné 2006), but has generated conflicts between industrial fishers and subsistence fishers (Metcalfe et al. 2016).

Artisanal fishing is a small scale activity where some of the catch is sold locally and the rest is kept to feed the family (King 2007). Most tropical countries in the world are developing countries where artisanal fishing occurs intensively in coastal waters and is essential for food security and to alleviate poverty (Allison and Ellis 2001; Béné 2006). However, the management of these artisanal fisheries is usually overlooked because fisheries policy is usually focused on industrialized fisheries (Allison and Ellis 2001). As a result, small-scale fisheries are poorly managed (Saavedra-Díaz et al. 2015). While small-scale fisheries are most common in the tropics, artisanal fishing also occurs in temperate waters, but at a smaller scale. However, conflicts with industrial fishers in temperate waters also arise because the same species are targeted despite the differences in technology (Frid et al. 2016). As a consequence, small-scale fisheries are continuously underappreciated, undermanaged and strongly affect fisher poverty alleviation and traditions.

1.3. Small-scale fishing in two distinct latitudinal places

The Ciénaga Grande de Santa Marta is located on the Caribbean coast of Colombia. It is the largest and most productive lagoon system in the country and is the fishing area for ~3500 fishers situated around the system in seven villages. Typically the catch consists of multiple species although three to four groups (Gerreids, Mugilids, shrimps and crabs) dominate the catch (Rueda et al. 2011). The human population lives under extreme poverty and with many basic unsatisfied needs, and fishing is their major food and income source (Vilardy et al. 2011). As such, fishing is an extremely important activity in this region because it helps to alleviate poverty to some extent. However, the ecosystem is under multiple anthropogenic stressors that deteriorate habitat and cause fish population declines coupled with high and unsustainable fishing pressure.

In British Columbia, Canada, on the other hand, First Nations communities along the coast have harvested marine resources for thousands of years. They continue to do so for traditional purposes and as a food source (Yamanaka and Logan 2010; Eckert et al.

2017). One of First Nations' preferred fish to harvest are rockfish (genus *Sebastes* spp.), particularly Yelloweye (*Sebastes ruberrimus*). However, industrial fishing has exerted a high fishing pressure on many rockfish species and, as a consequence, the fishery was overexploited and populations depleted (Yamanaka and Logan 2010). Furthermore, rockfish are particularly vulnerable to overfishing because their age at maturity occurs late in life (Love et al. 2002) making population recovery slow. Rockfish use habitats that are complex and in many cases this complexity is provided by biogenic structure such as sponges (Du Preez and Tunnicliffe 2011). Thus, their habitat is deteriorated by some fishing practices such as trawling.

In both the Ciénaga Grande de Santa Marta and waters of coastal BC artisanal fisheries are very important as a food source and as tradition. However, the resources these stakeholders depend on have been depleted by the combination of overexploitation and habitat destruction. Therefore, a deeper understanding on habitat use and distribution of species is critical in order to improve habitat conservation plans, fishery management and spatial planning.

The overall goal of this dissertation was to explore habitat use by fish (and some crustaceans) in mangrove-dominated systems in the tropics and in rocky reefs in temperate waters of BC at global, regional and local scales as a potential mechanism to help sustain small-scale fisheries. To achieve this goal I conducted different statistical techniques and fish surveys and included Local Ecological Knowledge approaches. Under the overall goal my main objectives where: i) Determine the mangrove-fishery relationship at a global scale conducting a meta-analysis. ii) Investigate the importance of mangrove habitats in a tropical lagoon system in the Colombian Caribbean. iii) Explore the mangrove-fishery linkage from a Local Ecological Knowledge approach in Ciénaga Grande de Santa Marta; and iv) Relate rockfish habitat characteristics) to performance of species distribution models at large spatial scales (>100 kms) in inshore waters of Vancouver Island, British Columbia.

In chapters 2-4, I focus on tropical mangrove-dominated systems to attempt to disentangle the importance of mangroves as fish habitat at a global and local scale, and with different methodological approaches. In Chapter 2, I present results of a meta-analysis of the mangrove-fishery relationship based on mangrove area and catches as previous reviews had only addressed the relationship from a qualitative perspective. After an extensive systematic literature review I extracted multiple data points from 23 papers. The analysis showed that the overall relationship between mangrove area and catches was positive and strong. Although I tested many moderators, the only significant moderator was the country where the studies were conducted. I suggest that these differences arise for two main reasons: mangrove diversity, productivity and abundance vary due to climatological, hydrodynamical and geomorphological settings that are more similar within a country. As a result, differences across countries occur. Alternatively, mangrove conservation and fishery management policies that differ between countries can explain the variation in effect sizes observed. Both scenarios, however, suggest that mangrove conservation must be a priority to protect critical fish habitats.

After demonstrating the strong mangrove-fishery linkage relationship at a global scale, in Chapter 3, I focused on how fish use different habitats in a lagoon system in the continental Caribbean because most studies conducted in this region have been done on islands where the seascape differs from lagoon systems. By collecting fish at different sites and distances from mangroves I tested whether fish abundance, diversity and number of immature fish were higher in mangrove habitats relative to habitats further from mangroves. Although fish use mangrove habitats, diversity and abundance decreased as a function of salinity while the number of immature fish slightly increased when salinity was higher, suggesting that salinity was the major driver of these variables in the system.

Although fishers are one of the main stakeholders in mangrove-dominated areas, they are rarely taken into consideration when conducting mangrove-fish studies. Thus, in Chapter 4, I examined the mangrove-fishery linkage from a novel perspective, using Local Ecological Knowledge. I used semi-structured interviews to determine fishing gear and species distributions, mangrove use, and fishers' perception of the importance of mangrove for fish and crustaceans. I found that mangroves provide more functions to fishers other than just food. Mangroves were used for firewood and as construction material. There is a general consensus among fishers that mangroves are essential to support fishing because fish and crustaceans use them as important habitats such as nurseries, spawning and feeding grounds.

Finally, in Chapter 5, I explored the importance of some benthic habitat characteristics for Rockfish (*Sebastes* spp.) in a temperate system by conducting species distribution models of both presence/absence and abundance at a large spatial extent (100s km). I used longline data of inshore waters on Vancouver Island collected by

Fisheries and Oceans Canada (DFO) over multiple years (2003 – 2016). I demonstrated that the prediction of Rockfish abundance at large scales is accurate and that patterns observed at small scales, such as complex rocky reef used by fish, hold at large scales and with medium resolution (20m) bathymetry data. These results suggest that complex habitat is important for rockfish and thus, these models should be accounted for when conducting spatial planning in British Columbia waters.

The results presented in this dissertation describe the importance of habitats and how fish (and some crustaceans) use habitat in tropical and temperate ecosystems. Habitat structural complexity (mangroves and rock) is critical for fish populations. However, most importantly the results are informative from a conservation and management perspective. Critical habitats should be protected in order to maintain fish production. However, holistic conservation plans that include whole ecosystems must be considered in order to enhance fish production by these habitats. Furthermore, including fishers' opinion in fishery management plans are required in order to decrease harvesting pressure on the populations. Finally, spatial planning based on species distributions and habitat use may be effective in order to recover fish populations.

Chapter 2 - Mangroves enhance local fisheries catches: A global meta-analysis

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2.1. Abstract

Mangroves are among the most productive ecosystems in tropical and subtropical regions. Historically, mangroves are assumed to support artisanal fisheries, leading decision makers to protect mangroves based on this premise. However this relationship remains unclear, despite positive correlations obtained in different geographical regions. Here, we provide the first meta-analysis of the mangroves-fisheries linkage at a global level. After conducting a systematic review, 23 publications containing 51 studies estimating the mangrove – fishery linkage were obtained. A random effect model was used to estimate the effect size (Pearson's correlation coefficient) of each individual study as well as the overall effect size. We found strong evidence for the mangrove-fishery linkage with an overall effect size of $r = 0.73$ (95% CI: 0.61 - 0.81) and substantial heterogeneity was observed ($Q = 143.88$, $df = 50$, $p < 0.01$). The countries where the studies were carried out was the only significant moderator ($Q_M = 26.07$, $p < 0.01$) while fisheries types (*i.e.* crab, fish, shellfish, prawn and total) and global regions were not good predictors of the relationship. Our results show that mangrove area is a good predictor of fishery catches overall, confirming the importance of conserving such habitats.

2.2. Introduction

Mangrove forests support a high diversity of both marine and terrestrial fauna from a variety of taxonomic groups that carry out critical ecosystem functions

(Kathiresan and Bingham 2001; Nagelkerken et al. 2008). Despite their importance mangroves are being lost at alarming rates (Valiela et al. 2001). One of the most studied mangrove ecosystem functions is the mangrove-fishery linkage based on the role of mangroves as nurseries for marine and estuarine species. Many species that use mangroves undergo ontogenetic habitat shifts from mangroves to adjacent ecosystems (*e.g.* coral reefs, soft bottoms, pelagic ecosystems) (Nagelkerken et al. 2001; Mumby et al. 2004). However mangrove nursery function varies at local scales where tide dynamics, turbidity and geomorphological settings differ (Castellanos-Galindo and Krumme 2013). Three hypotheses have been proposed to support the nursery concept, i) high food availability (Laegdsgaard and Johnson 2001) ii) protection from predators and iii) shelter from a number of physical disturbances (Manson et al. 2005b). Habitat complexity provided by prop roots and pneumatophores decrease predator-prey encounters, which in turn decrease predation risk for juveniles using mangrove habitats (Laegdsgaard and Johnson 2001). However, decreased visibility caused by high turbidity and shallow waters are alternative mechanisms decreasing predation risk by reducing visibility and avoid large predators from entering these habitats(Primavera 1997; Beck et al. 2001; Nagelkerken 2009). While the nursery concept has been well documented in the Caribbean (Mumby et al. 2004) it has been difficult to assess in the Indo West Pacific (IWP) (Lee et al. 2014). However, recent otolith stable isotope analysis suggests the importance of mangroves as nurseries in the IWP (Kimirei et al. 2013). While such evidence increases (Igulu et al. 2014), further research is needed to determine recruit movement, growth and survival from mangrove areas to adult habitats (Beck et al. 2001; Nagelkerken 2009).

To date the mangrove-fishery linkage remains controversial. For decades mangroves have been assumed to be important nursery areas for commercially important species such as Penaeid shrimp and fish, suggesting a direct and positive relationship between mangrove area and both coastal and offshore fisheries. While this assumption has contributed greatly to the development of conservation programs and strategies to protect mangroves that have favoured the ecosystems goods and services provided by them, the ecological aspects of the fishery linkage are still poorly understood (Manson et al. 2005b). The first studies to quantify these relationships carried out a simple linear regression approach where catches were regressed against mangrove area (Martosurbroto and Naamin 1977). As new and more powerful statistical methods became available more explanatory variables (e.g. freshwater discharge, coastal length, estuarine area) were included in the models to better understand this relationship (Loneragan et al. 2005; Meynecke et al. 2007). However, there are still differing opinions about whether mangroves are a good predictor of fisheries production. For example, Lee (2004) analysed the relationship between mangrove abundance and prawn production worldwide and concluded that the extent of intertidal areas explained variability in prawn production better than mangrove area. In contrast, other studies have found positive correlations between mangroves and fisheries in different geographic settings and scales (Manson et al. 2005a; Aburto-Oropeza et al. 2008; Carrasquilla-Henao et al. 2013).

Previous reviews have discussed the mangrove-fishery linkage (Manson et al. 2005b; Blaber 2007). Although these reviews have oriented researchers towards new and important research directions, to our knowledge there has not been a quantitative analysis approach to the mangroves-fisheries linkage at a global level. We conducted a random effect meta-analysis of the mangrove-fisheries linkage relationship, to determine i) whether mangroves are good predictors of fishery catches and ii) if there are global patterns in the relationship between mangroves and associated fisheries.

2.3. Methods

2.3.1. Data collection

We conducted a systematic review on Thomson Reuters' Web of Knowledge database. The terms, mangrove* AND fisher* OR mangrove* AND prawn* were used and 1663 hits were obtained. Fisher* rather than fish* was chosen to narrow our search to studies that addressed mangrove-fishery relationship and not other measurement of abundance which have been conducted elsewhere (*e.g.* Serafy *et al.* 2015). 1166 hits were discarded based on the title and abstract as not relevant for the analysis. 497 hits were carefully analyzed for possible inclusion. For a paper to be included in the analysis it had to meet three criteria; first, the paper should be a study of organisms (fisheries) that was conducted in mangrove habitats. Second, a relationship between the organism studied and mangrove coverage had to be mentioned. Third, evidence that the catch data of the organism(s) was related to mangrove (*e.g.* used mangroves as habitat, feeding ground or nursery) should be provided. Finally a statistical relationship (*i.e.* correlation, regression or r^2 of the model) between mangrove area or perimeter and catches or enough information to calculate it had to be presented (Fig. 2.1). Peer reviewed papers and governmental reports were included in the analysis. Although most of the extraction was from the Web of Knowledge some publications (seven) were found by following the references cited in the papers (where governmental reports were found). In total we included 23 publications and 51 studies in our analysis. Here, study is defined as each individual relationship between mangroves and catches, as many papers show the relationship for more than one fishery for the same mangrove area (*e.g.* Paw and Chua 1991; Manson *et al.* 2005; Carrasquilla-Henao *et al.* 2013).

2.3.2. Statistical analysis

We extracted the correlation coefficient (r), coefficient of determination (r^2) or adjusted R^2 and sample size (n) for the relationship between mangrove area or extent and catches for each fishery. Correlations and regressions are widely used in ecology to explain relationships between continuous variables, which has led to several meta-analysis studies using this parameter as an effect size (Harrison 2011). Some studies (*e.g.* Koricheva 2002) suggest calculating the square root of the coefficient of determination to obtain r. However, this calculation is biased especially when sample sizes are small (Nakagawa and Cuthill 2007). Thus, for all the studies that provided a coefficient of determination, r^2 or an adjusted r^2, , we calculated an $r_{adjusted}$ based on the equation provided by Nakagawa and Cuthill (2007):

$$r_{adjusted} = \sqrt{1 - \frac{(n-1)(1-R^2)}{n-k-1}}$$

where n is the sample size and k is the number of explanatory variables. One included study (Ley 2005) calculated Spearman's rank correlation so it was converted to Pearson's correlation coefficient (r) by

$$r = 2\sin(\frac{\pi\rho}{6})$$

where ρ is the rank correlation (Lajeunesse 2013).

Next, we transformed each value of r in order to follow a normal distribution given that values close to ±1 are skewed (Viechtbauer 2010) with Fisher's z transformation:

$$z = \frac{1}{2}[\ln\frac{(1+r)}{(1-r)}]$$

From the relationships we categorized the fisheries into five groups, fish, prawn, crab, shellfish and total. This was done because some studies did not specify the species used in the analysis, and to increase the sample size given that species-specific studies were scarce. The 'total' category is used for those studies that related the total catch (i.e. two or more groups together) to mangroves. Three regions were used to globally locate each study, i) IWP ii) Americas, Caribbean and Eastern Pacific (ACEP) and iii) worldwide for those studies that conducted an analysis in more than one region. Finally, the country where each study was carried out was recorded in the database as an additional moderator.

An overall effect size of the relationship between mangrove area and fisheries was calculated with a random effect model. This model was used because the studies included were not all identical. They differed in location, mangrove species composition, tidal regime and fishery caught, among many others. Also, we wanted to draw conclusions to the entire population of mangroves – fishery studies and not limit it to the 23 publications included in the present study (Viechtbauer 2010). Moreover, we conducted a multilevel (three level) meta-analysis to account for the nested structure of our dataset, that is, studies nested within publications. This framework allowed us to account for the non-independence of the individual correlations extracted from publications with more than one mangrove-fishery relationship (Nakagawa and Santos 2012). Thus, the variance obtained by the three levels are: within study variance (level 1), between studies within same publication variance (level 2), and between publications variance (level 3) (Konstantopoulos 2011). In summary, we included the effect sizes nested within publications as a random effect in our model. After calculating the overall effect size, mixed effect models were conducted to determine whether some categorical moderators (fishery, region, and/or country) explained at least part of the heterogeneity. Omnibus tests were carried out to test whether the coefficients of the model were significant and if the moderator itself was significant (Viechtbauer 2010). A full model including all moderators was conducted followed by a number of different models including different combinations of the moderators. The best model was chosen based on Akaike Information Criterion (AIC) scores.

The heterogeneity of the overall model and the mixed effect models were calculated with the σ^2 parameter, while the heterogeneity significance of each model was tested with Cochran's Q test (Borenstein et al. 2009; Favaro and Côté 2015). When including moderators in a model the total heterogeneity in the model (Q_T) is portioned into Q_M, the heterogeneity explained by the moderators, and the residual or unexplained heterogeneity (Q_E). Thus, the total heterogeneity (Q_T) is the sum of the unexplained variability (Q_E) and the variability explained by the covariate (Q_M) (Borenstein et al. 2009). All the analyses were carried out with R version 3.1.2 (R Core Team 2014) and the meta-analysis was conducted with the Metaphor package (Viechtbauer 2010).

A major concern when conducting a meta-analysis is publication bias that occurs when some studies may be excluded from the analysis because they have not been published. Thus, a difference in the results between unpublished and published studies exists (Møller and Jennions 2001). We explored publication bias by plotting a funnel plot of the overall effect model. If no publication bias is present the estimated effect size for each study should be closer to the overall estimated effect size as sample size increases (*i.e.* symmetrical) (Santos et al. 2011).

In addition to the visual exploration we conducted a fail-safe number tests to determine whether publication bias was present. The fail-safe number estimates the number of non significant studies that must be included to make the overall effect non-significant (Harrison 2011). We then compared the number obtained to the number obtained by Rosenthal's method:

$$n = 5k + 10$$

where *k* is the number of studies in the meta-analysis. If *n* is smaller than the fail-safe number then the results are robust (Jennions et al. 2013).

Lastly, we tested for temporal trends in our dataset by conducting a cumulative meta-analysis to determine whether the magnitude of the effect sizes of the mangrove-fishery relationships have changed over time as new evidence, statistical methods and study sites have developed. Cumulative meta-analysis accounts for the temporal trend in the mean effect size by adding one study at a time and recalculating the overall effect size and confidence intervals (Leimu and Koricheva 2004). We organized our studies in chronological order from oldest to most recent when conducting the analysis. We also conducted a mixed model using year as a moderator to obtain more robust temporal results.

2.4. Results

Our search terms produced a total of 506 publications of which 23 publications from both peer reviewed and grey literature yielded 51 independent data points, 11 of which were from the ACEP region, 37 from the IWP and only three worldwide studies. In the ACEP and IWP regions prawn and fish were the most common fisheries used to study linkages between mangrove area and landings whereas only prawns have been tested on a global scale (Fig. 2.2).

The effect sizes from the individual studies ranged from $r = -0.56$ to $r = 0.98$ (Table 2.1). The overall effect size of the mangrove and fishery relationship among all studies estimated from the random effect model was $r = 0.72$, (95%CI: 0.61 - 0.81) thus significantly different from 0 (Fig. 2.3). The variability between publications was larger ($\sigma^2 = 0.18$) than studies within publication ($\sigma^2 = 0.011$) The random effect model also suggested substantial heterogeneity between studies ($Q_T = 143.88$, df = 50, p < 0.0001).

Based on the AIC scores the moderator that best explained the variability in the relationship was "Country" (Table 2.2). Moreover, when including regions ($Q_E = 129.1$, df = 48, p < 0.001) and fisheries ($Q_E = 127.42$, df = 46, p < 0.001) as moderators substantial heterogeneity persisted with an extremely low percentage of heterogeneity accounted for in the model. The omnibus tests in both cases suggested that these moderators do not influence the relationship between mangroves and fisheries ($Q_M = 2.10$, df = 2, p = 0.35 and $Q_M = 4.38$, df = 4, p = 0.36 respectively) even though the levels of the factors were significantly different from 0 (Fig. 2.4a and 2.4b).

When including the moderator country in a mixed effects model considerable heterogeneity remained unexplained ($Q_E = 89.46$, df = 44, p < 0.0001). However 68.9% of the total amount of heterogeneity could be accounted for with this moderator ($Q_M = 26.07$, df = 6, p < 0.01). The omnibus test suggests that the coefficients are different from 0, indeed the estimated correlation coefficient of the effect of mangrove area on catches for all countries was positive and significant (Fig. 2.3) ranging from $\beta_7 = 0.49$ (worldwide) to $\beta_8 = 0.98$ (Vietnam).

A visual inspection of the relationship between the correlation coefficient and sample size suggests a funnel shape. That is, there is higher variability in the individual correlation coefficient at small sample sizes and it narrows as sample size increases (Fig.

2.5). The fail-safe number estimated to make the results non-significant was 7165, which also suggests no publication bias. Rosenthal's method number was 265, well below the fail-safe number, 7165. Therefore, we confirmed the robustness of our results.

The cumulative meta-analysis shows consistency in the magnitude of the effect size through time (Fig 2.6). Although the confidence intervals were wider in the earlier studies these did not overlap the non-significance correlation (*i.e.* $r = 0$). The confidence intervals obtained in the analysis tended to narrow down from 2004 to 2015 suggesting a strong relationship in the mangrove-fishery linkage. The mixed model using year as moderator yielded similar results. "Year" was not a good predictor of the mangrove-fishery linkage ($Q_M = 0.64$, df = 1, $p = 0.42$). These results suggest that throughout the 38 years that this relationship has been studied the outcomes have been consistent.

2.5. Discussion

Most previous studies trying to determine whether mangroves support fisheries have been approached qualitatively (Baran 1999; Manson et al. 2005b; Blaber 2007). However, a recent study has gone beyond this this qualitatively approached and developed a mangrove –fishery model based on expert judgment whereby nutrient and freshwater input and mangrove area where predictors of potential fish catch. Although this study has gone a step forward the model is yet to be parameterized with local catch data (Hutchison et al. 2015). Our study is the first attempt to quantify the mangrove-fishery linkage globally. Meta-analysis has a number of advantages with respect to qualitative reviews or vote counting procedures. While reviews provide expert opinion they may be subjective, and vote counting suffers from poor statistical procedures, meta-analysis offers a set of statistical tools to quantify the overall outcome of controversial ecological questions (Koricheva and Gurevitch 2013). Results of our meta-analysis suggest that mangroves have a strong effect (*r* = 0.72) on fisheries in a variety of mangrove settings across the world. One of the major criticisms that this result faces is that correlation does not imply causation (Blaber 2009; Lee et al. 2014). Although this statement is true, and is a limitation of correlation meta-analyses (Worm and Myers 2003), our understanding of the function of mangrove as nurseries in different mangrove settings has increased over the past few years (Kimirei et al. 2013; Igulu et al. 2014). However, further studies in different mangrove settings are still needed. While we consider that one of the most important mechanisms driving the relationship between mangroves and fisheries is the importance of mangroves as nursery habitats for commercially important species we acknowledge that juveniles of different species can utilize mangroves in different ways such as a sources of food, shelter or both (Nagelkerken 2009). Nonetheless, a recent study conducted by Serafy et al. (2015) in the Wider Caribbean Area demonstrated that between 6 and 8 mangrove dependent fishes' abundance is proportional to mangrove area and that other predictors such as latitude and population density are not as strong predictors as mangrove area. Similarly, Igulu et al. (2014) found that for some species juvenile fish density was higher in mangrove habitats than in adjacent habitats in both IWP and the Caribbean. Earlier studies have also found that juvenile fish densities are higher in mangrove habitats compared with unstructured habitats such as mudflats in the IWP (*e.g* Robertson and Duke 1987; Chong *et al.* 1990) and the Caribbean (Nagelkerken and Van der Velde 2002). These findings suggest that when mangrove habitats are present fish density is higher in these systems compared to adjacent habitats probably due to the higher habitat complexity provided by mangroves. While nursery studies in the Caribbean have focused on island mangroves that lack freshwater discharges and have seagrass beds and coral reefs as adjacent habitats (*e.g.* Mumby *et al*. 2004), different mangrove settings occur in the Caribbean such as lagoon systems (*e.g.* Cienaga Grande de Santa Marta in Colombia), that have different adjacent habitats and completely different dynamics that thus far, have not been studied. Evaluating the importance of mangroves as nurseries for many species in mangrove estuaries in the Caribbean may provide additional knowledge in the field. Also, more mangrove habitats with varying environmental characteristics (*e.g.* tide regimes and precipitations) must be researched across the globe to fully explain the importance of mangrove as nurseries (Castellanos-Galindo and Krumme 2013) and therefore fill the knowledge gaps that remain in the mangrove-fishery linkage, to date, highly absent in the ACEP region.

We conducted a multi-level random effects model as we accounted for the nested structure of our data and anticipated considerable heterogeneity among studies given their differences in geographical location and local environmental and geomorphological mangrove settings in addition to variability in the mangrove area. While the variability was higher across publications than within studies in a publication we were able to make inferences from a number of moderators given the multi-level framework approach used. The model yielded high variability and thus we used some moderators to try to explain such variance. We hypothesized that the regions (*i.e.* ACEP, IWP and Worldwide) would explain at least part of the heterogeneity because of their differences in species richness and geomorphological characteristics (Spalding et al. 2010; Lee et al. 2014) based on the hypothesis that productivity is enhanced by diversity (Tilman et al. 1996). That is, that the higher mangrove diversity in the IWP would provide higher productivity and thus increased catches. Furthermore, a strong relationship between mangrove species richness and sponge, brachyuran crabs, and gastropod richness has been shown at a global scale (Ellison 2008). However, we found that region was not a good explanatory variable despite the positive correlations observed. This can be caused by a number of mechanisms; although the IWP is substantially higher in mangrove species richness, and no overlap in species among regions exists, the architectural structure of pneumatophores and aerial roots are alike in similar species (*e.g. Rhizophora mangle* vs. *Rhizophora apiculata* and *Avicennia germinans* vs. *Avicennia marina*). These structures are found in both realms and may be providing habitat complexity analogues despite the biogeographic differences across regions. Also, the ACEP is underrepresented with only one country, Mexico. Despite having studies carried out in both the Gulf of Mexico and the Gulf of California the few studies in the ACEP region can also explain the lack of heterogeneity explained by the "regions" moderator. Indeed, the model parameterization, Y_i ~ region + country, was excluded from the AIC table because the model drops redundant parameters. This implies that it drops "Mexico" from a moderator and thus the outcome is the same as for "countries" by itself. Moreover, in a meta-analysis conducted by Igulu et al. (2014) the authors found that tidal regimes are a better predictor of nursery habitats than the regions themselves. However, Lee et al. (2014) suggest that mangrove ecosystem function may differ across regions given the presence or absence of different key species in both realms.

Unlike regions, country explained about 70% of the heterogeneity in the mangrove-fisheries linkage model. A probable explanation is that mangrove diversity varies at continental and regional scales in response to environmental factors such as precipitation, and interspecific and intraspecific competition (Ellison and Farnsworth 2001). Alternatively, mangroves differ in size, productivity and abundance at specific locations driven by precipitation, climatological conditions, tidal regimes, freshwater flow and geomorphological and edaphic conditions (Duke et al. 1998; Alongi 2009; Castellanos-Galindo et al. 2013; Hutchison et al. 2013). Thus, mangrove habitats are likely more similar within countries. Hence, countries accounted for the bulk of the variability in the model. Another possible reason why countries was such a good predictor is that there are consistent differences among countries with respect to mangrove conservation and fishery management policies that can change the magnitude of the correlation. However, this does not apply to the 'worldwide' region as this factor includes studies conducted in more than one country where conservation management plans likely differ. We collected data for six different countries (Australia, Malaysia, Indonesia, Philippines, Vietnam and Mexico). According to Spalding et al. (2010) 125 countries in the world have mangroves therefore the proportion of countries that have conducted these studies is low (4.8%). Historically, research in the IWP has focused on trying to disentangle the mangrove-fisheries linkage while in the ACEP, particularly in the Caribbean, research has focused on understanding mangroves' nursery function for coral reefs (*e.g.* Nagelkerken et al. 2001; Mumby et al. 2004; Igulu et al. 2014). Thus, future mangrove-fisheries linkage studies should be carried out in Africa, Central America and the east and west coasts of South America.

Historically, prawn fisheries linkages to mangroves have received much more attention (40%) than other fisheries, especially in the IWP (Fig. 2.2). The five categories of fisheries included in our model did not explain any of the heterogeneity even though most of them, except for shellfish, showed positive correlations. These categories were binned into larger groups that included more than one species. For example, 'fish' was a broad category that contained a number of different species such as mullets, snappers, groupers and mackerels among others. Similarly, 'prawns' included both banana and tiger prawns. This approach was necessary because several studies included in the analysis classified the fishery used in their relationship in a broad group, thus hampering our ability to precisely classify them. Different species and different taxonomic groups use habitats in different ways and at different spatial and temporal scales therefore the relationships are likely to differ across groups. Despite the lack of variability accounted for by the model for this moderator, species-specific studies on prawns have demonstrated the importance of mangroves for their life cycles (Ronnback et al. 2002; Vance et al. 2002). Similarly, shellfish (*i.e.* bivalves) are usually harvested by hand from mangrove mud sediments (Mackenzie 2001). While it is evident that mangrove area is important for such groups it was not significant in the model. However, there were only two studies that provided evidence on the relationship between shellfish and mangrove area thus hampering the possibility of observing a potential strong relationship. Nonetheless, relationships between mangrove area and non commercially important species' richness has also been shown at global scales (Ellison 2008). Thus, it is possible that if species relationships to mangroves were considered individually the model could account for some of this variability as has been shown for some reef species in the wider Caribbean (Serafy et al. 2015). Therefore, future studies should focus on species-specific relationships, however from a fisheries perspective, this may be challenging because i)most local fishery offices do not report catches at a species level and ii) catches in the tropics and subtropics are highly diverse (Blaber 2007).

The publication bias tests we conducted yielded compelling results with respect to the mangrove-fisheries linkage. The fail-safe number test suggests that our study does not suffer from publication bias. In fact, because we extracted some studies from the grey literature we reduced a major problem in meta-analysis, publication bias. When a new research avenue opens the first studies to be published usually are significant, however the outcomes tend to vary through time (Santos et al. 2011). In our cumulative meta-analysis (Fig. 2.6) this trend is clearly observed where the initial studies presented strong correlations but wide confidence intervals. As time progresses the confidence intervals decreased but the magnitude of the relationship remained consistent. Although strong correlations remain in recent studies (*e.g.* Arbuto-Oropeza *et al*. 2008; Carrasquilla-Henao *et al*. 2013; Vázquez-González *et al.* 2015), the overall outcomes differ by author, fishery and region. Although recent studies have increased the number of predictors included in the models leading to more variable outcomes little change in the overall effect was observed. While our temporal results provide compelling evidence on the importance of mangrove area as a predictor of catches we understand that this predictor must not be accounted for independently rather, many other explanatory variables (e.g., size of estuary, freshwater flow, salinity etc.) should be used together with mangrove area. This will contribute to increasing evidence of connectivity among different habitats and will expand our coastal shallow ecosystem seascape understanding (Nagelkerken et al. 2015). In studies that included several explanatory variables (*e.g.* Lee 2004; Loneragan *et al*. 2005; Meynecke *et al*. 2007) the importance of mangroves tended to be slightly weaker than in studies that only included area as a predictor, however the meta-analysis outcome still yielded an overall strong correlation. The temporal analysis does not suggest a decrease in the strength of the correlation through time despite greater variability in the studies' outcomes in recent years.

The result of our meta-analysis has important conservation implications. Although overfishing, pollution, and land cover change have detrimental consequences for marine ecosystems (Lotze et al. 2006; Halpern et al. 2008), mangrove forest degradation can substantially contribute to catch declines by the removal of critical habitats used by many commercially important species. Despite the importance of mangroves as fish habitat and the number of ecosystem functions and ecosystem services they provide, they are being lost at alarmingly fast rates (Valiela et al. 2001; Alongi 2002; Spalding et al. 2010), adding to the many other problems coastal ecosystems are facing. Our analysis of the mangrove-fisheries linkage suggests that globally mangroves have a considerable effect on fisheries. The effect seems to be more similar within countries as opposed to regions or fishery, which highlights the importance of local conservation strategies and strong governmental policies to protect mangroves as critical habitats and as an important food source for vulnerable families in developing countries.

Table 2.1. Pearson's correlation coefficients calculated from the random effect model and back transformed (atanh (r_z)) for each study and organized by fishery (Crab, Fish, Prawn, Shellfish and Total). The sample size of each study is shown together with the Country and region (IWP = Indowest Pacific, ACEP = Atlantic Caribbean and Eastern Pacific, Worldwide = studies conducted in more than one country) where they were conducted.

Authors	Fishery	Pearson's correlation coefficient (r)	Sample size (n)	Country	Region
Manson *et al*. 2005	Crab	0.71	36	Australia	IWP
Carrasquilla *et al*. 2013	Crab	0.72	5	Mexico	ACEP
Meynecke *et al*. 2007	Crab	0.62	13	Australia	IWP
Meynecke *et al.* 2007	Crab	0.63	13	Australia	IWP
Jothy 1984	Crab	0.60	10	Malaysia	IWP
Yañez-Arancibia 1985	Fish	0.69	10	Mexico	ACEP
Paw and Chua 1991	Fish	0.63	20	Philippines	IWP
Paw and Chua 1991	Fish	0.73	18	Philippines	IWP
Paw and Chua 1991	Fish	0.81	12	Philippines	IWP
Paw and Chua 1991	Fish	0.63	18	Philippines	IWP
Paw and Chua 1991	Fish	0.58	15	Philippines	IWP
Saintilian 2004	Fish	0.72	17	Australia	IWP
Saintilian 2004	Fish	0.42	17	Australia	IWP
Manson et al 2005	Fish	0.56	36	Australia	IWP
Carrasquilla *et al*. 2013	Fish	0.71	5	Mexico	ACEP

Carrasquilla *et al.* 2013	Fish	-0.55	5	Mexico	ACEP
Meynecke *et al.* 2007	Fish	0.97	11	Australia	IWP
Ley 2005	Fish	0.29	11	Australia	IWP
Turner 1977	Prawn	0.76	21		World Wide
Matosubroto and Naamin 1977	Prawn	0.89	7	Indonesia	IWP
Staples *et al*. 1985	Prawn	0.76	6	Australia	IWP
Pauly and Ingles 1986	Prawn	0.51	38		World Wide
Sasekumar and Chong 1987	Prawn	0.94	10	Malaysia	IWP
Paw and Chua 1991	Prawn	0.78	18	Philippines	IWP
Paw and Chua 1991	Prawn	0.81	18	Philippines	IWP
Lee 2004	Prawn	0.38	37		World Wide
Loneragan *et al*. 2005	Prawn	0.75	8	Malaysia	IWP
Loneragan *et al*. 2005	Prawn	0.75	8	Malaysia	IWP
Loneragan *et al*. 2005	Prawn	0.21	8	Malaysia	IWP
Loneragan *et al*. 2005	Prawn	0.71	8	Malaysia	IWP
Manson *et al.* 2005	Prawn	0.80	36	Australia	IWP
Barbier and Strand 1997	Prawn	0.80	11	Mexico	IWP
Carrasquilla *et al*. 2013	Prawn	0.95	5	Mexico	ACEP
Meynecke *et al*. 2007	Prawn	0.80	13	Australia	IWP
Gedney *et al*. 1982	Prawn	0.90	11	Malaysia	IWP
Kenyon *et al*. 2004	Prawn	0.00	11	Australia	IWP
Kenyon *et al*. 2004	Prawn	-0.54	11	Australia	IWP

Kenyon *et al*. 2004	Prawn	0.00	11	Australia	IWP
Kenyon *et al*. 2004	Prawn	-0.56	11	Australia	IWP
Jothy 1984	Prawn	0.82	10	Malaysia	IWP
Sheaves *et al*. 2012	Prawn	0.70	25	Australia	IWP
Carrasquilla *et al*. 2013	Shellfish	0.73	5	Mexico	ACEP
Jothy 1984	Shellfish	0.68	8	Malaysia	IWP
Paw and Chua 1991	Total	0.63	34	Philippines	IWP
de Graaf and xuan 1998	Total	0.94	5	Vietnam	IWP
de Graaf and xuan 1998	Total	0.99	18	Vietnam	IWP
Aburto-Oropeza *et al*. 2008	Total	0.82	13	Mexico	ACEP
Aburto-Oropeza *et al*. 2008	Total	0.86	13	Mexico	ACEP
Carrasquilla *et al*. 2013	Total	0.96	5	Mexico	ACEP
Camacho and Bagarinao 1987	Total	0.72	60	Philippines	IWP
Vazquez-González et al. 2015	Total	0.81	99	Mexico	ACEP

Table 2.2. Akaike Information Criterion (AIC) for a combination of models depicting the relationship of the mangrove-fishery linkage effect size (Y_i) with respect to the moderators accounted for in the study. The AIC outcomes are organized from top (best) to bottom (less suitable). Larger ΔAIC and less weight represent less suitable models while heterogeneity is the explained variability by each model.

Model	*Δ AIC*	*Weight*	Heterogeneity σ^2
Y_i ~ country	0	0.46	0.69
Y_i ~ fishery + country	5.15	0.035	0.12
Y_i ~ region + fishery + country	5.15	0.035	0.61
Y_i ~ region	7.88	0.009	0.69
Y_i ~ fishery	9.26	0.004	0.05
Y_i ~ region + fishery	10.87	0.002	0.16

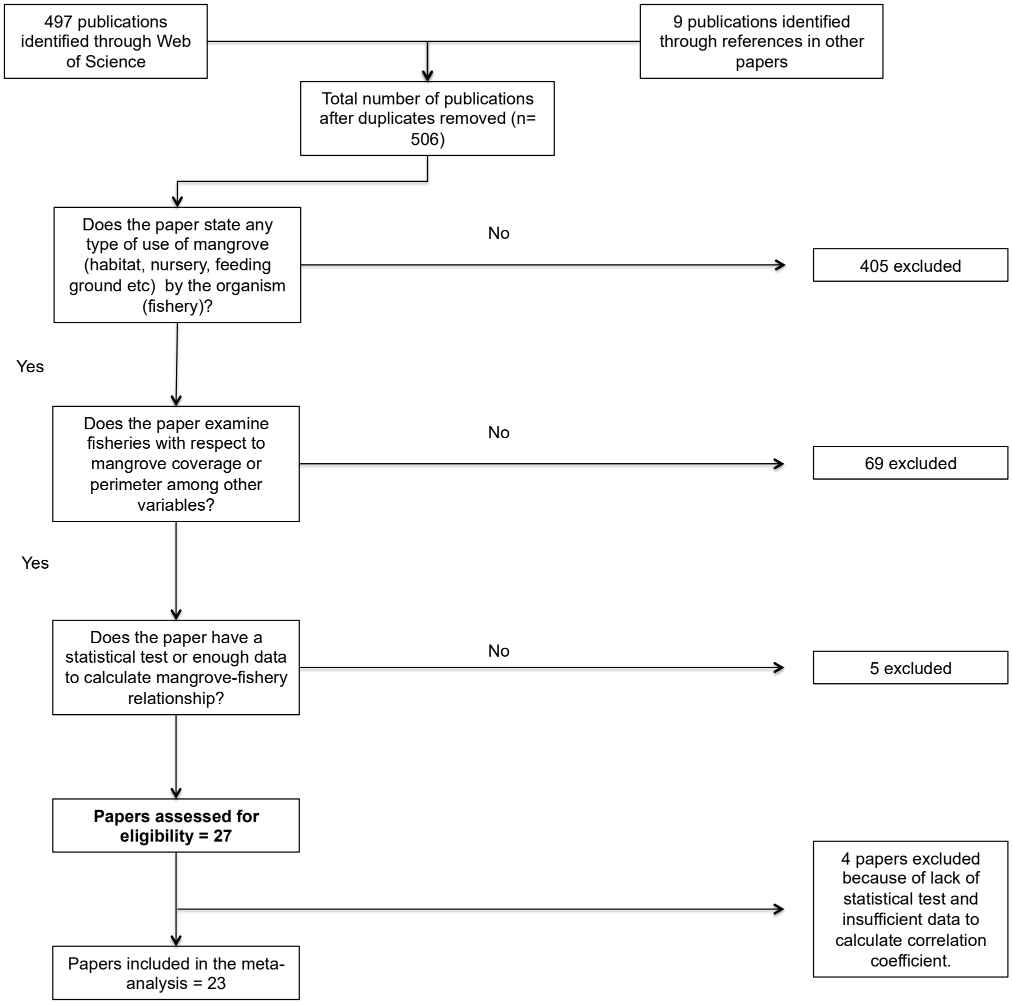

Figure 2.1. Decision making flow chart of the publications included in the analysis based on the Preferred Reporting Items for Systematic Reviews and Meta-Analysis (PRISMA statement) (Moher et al. 2009).

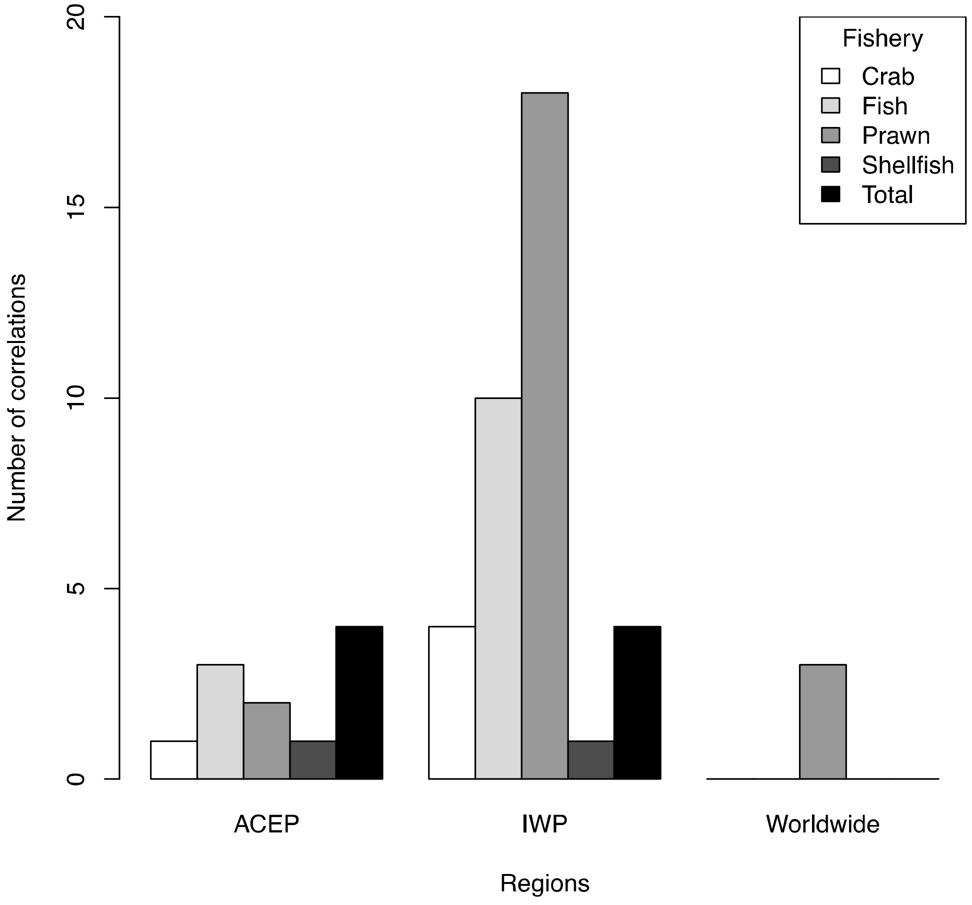

Figure 2.2. Correlation coefficient frequencies of the effect of mangrove area on catches in the three different regions. ACEP = American Caribbean and Eastern Pacific, IWP = Indowest Pacific and Worldwide.

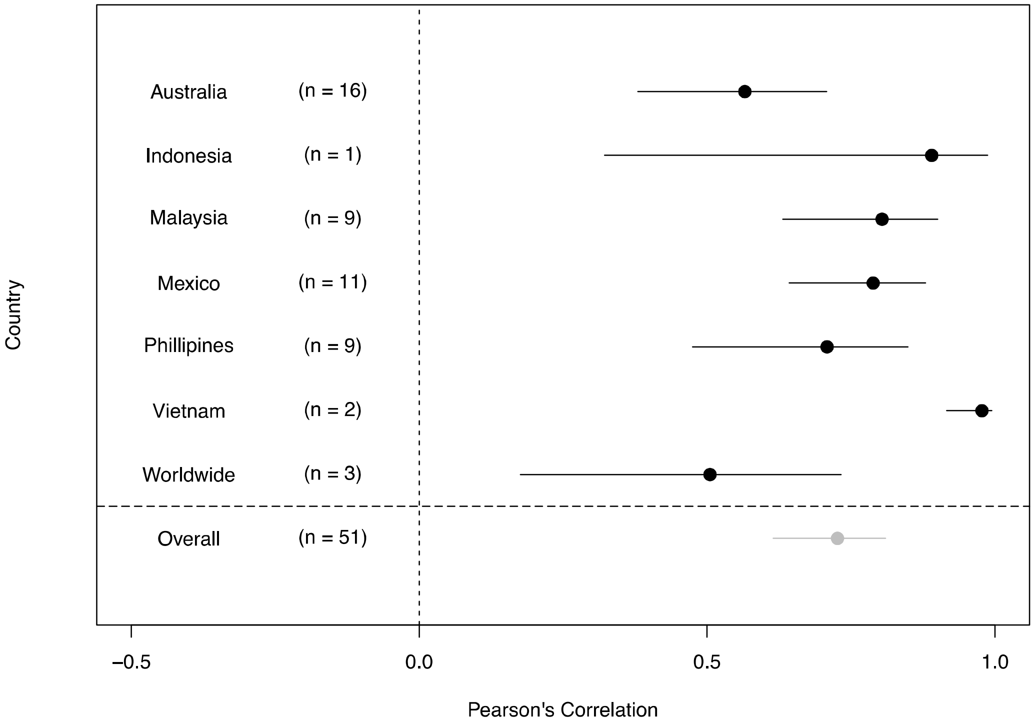

Figure 2.3. Forest plot showing the strength of the mangrove area – fishery relationship for different countries. Black points represent the modeled effect size by a mixed effect model for each country while the grey point represents the overall calculated effect size by a random effect model. Lines adjacent to the points represent the 95% confidence intervals. Vertical r line ($r = 0$) signifies no correlation while every thing to the right represents a positive correlation and on the left a negative correlation. When bars cross the no effect line (r =0) the estimated effect size is not significant.

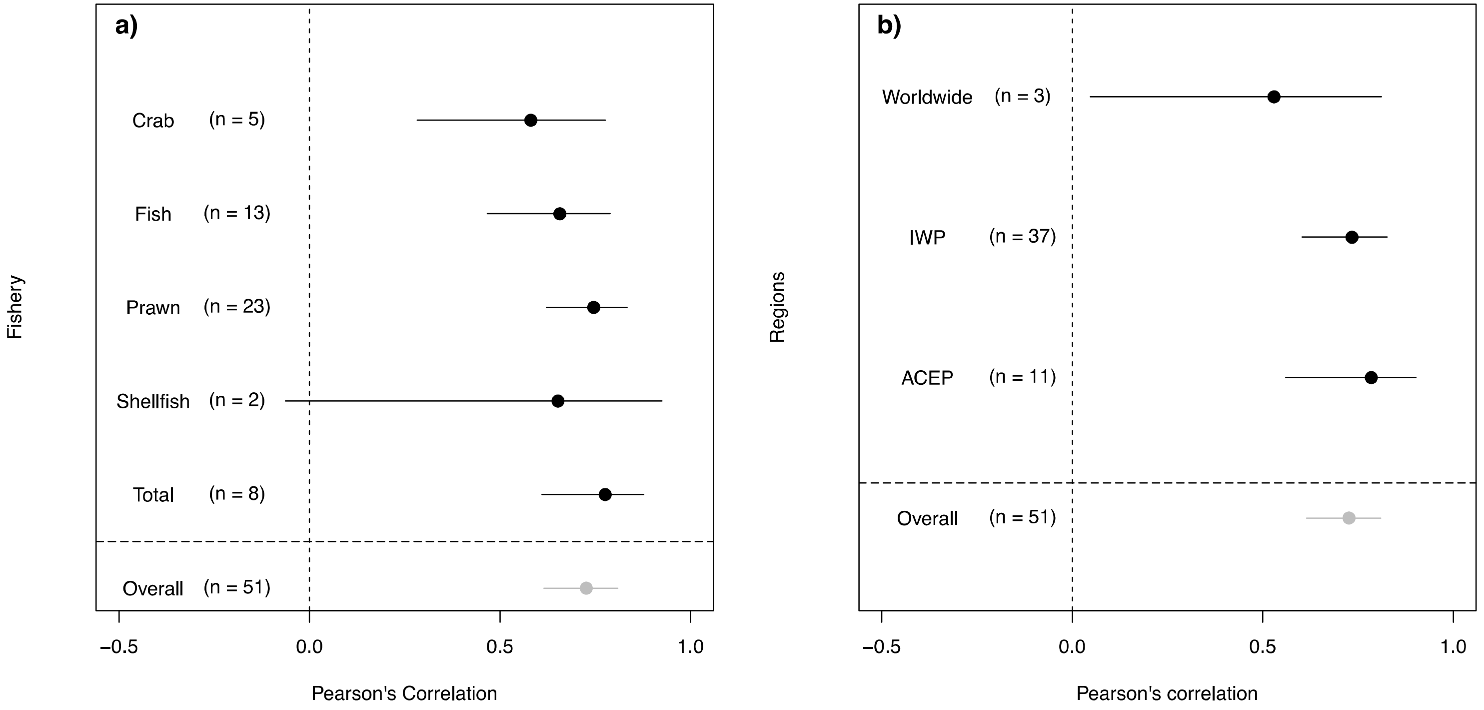
a)
Fishery
Crab (n = 5)
Fish (n = 13)
Prawn (n = 23)
Shellfish (n = 2)
Total (n = 8)
Overall (n = 51)
−0.5
0.0
0.5
1.0
Pearson's Correlation
b)
Regions
Worldwide (n = 3)
IWP (n = 37)
ACEP (n = 11)
Overall (n = 51)
−0.5
0.0
0.5
1.0
Pearson's correlation

Figure 2.4. Forest plot showing the strength of the mangrove area – fishery relationship for different a) fisheries and b)regions. The three regions are; (i) Worldwide (ii) Indo West Pacific (IWP) and (iii) Atlantic-Caribbean and Eastern Pacific (ACEP). Black points represent the modeled effect size using a mixed effect model for each a) fishery and b) region while the gray point represents the overall calculated effect size using a random effect model. Lines adjacent to the points represent the 95% confidence intervals. The vertical dashed line ($r = 0$) signifies no correlation with everything to the right representing a positive correlation and on the left a negative correlation. When bars cross the no effect line ($r =0$) the estimated effect size is not significant.

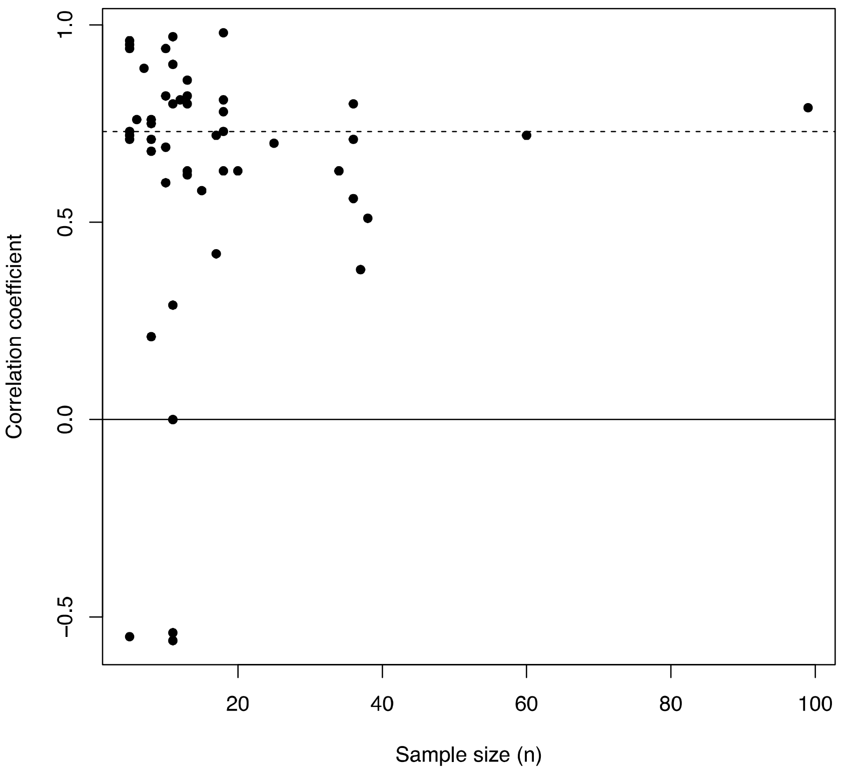

Figure 2.5. Funnel plots showing the relationship between Pearson's correlation coefficient (*r*) and sample sizes for all the studies included in the analysis. The dotted line represents the overall effect size calculated from the random effect model and the solid line represents no effect ($r = 0$).

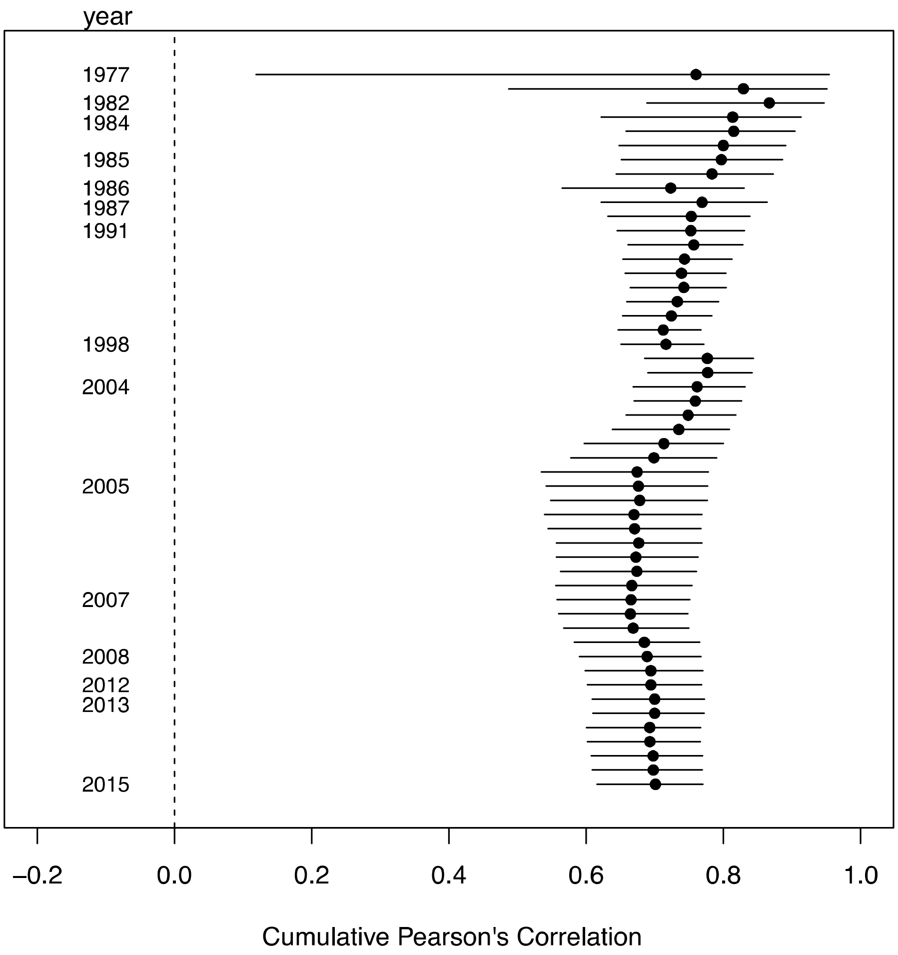

Figure 2.6. Forest plot showing a temporal (publication year) cumulative meta-analysis of the effect of mangrove area on different fisheries across the world. Where there is more than one publication for year the publication year is only shown in the first point. Dotted vertical line represents no effect (r = 0). Lines adjacent to the points represent 95% confidence intervals.

Chapter 3 - Evaluating mangrove habitat use by fish in a tropical Caribbean lagoon system.

Adapted from: Mauricio Carrasquilla-Henao[1], Mario Rueda[2] and Francis Juanes[1]. *Marine Ecology Progress Series. In Review.*

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Author contributions: M.C. and F.J. conceived of and designed the study. M.R. provided some data. M.C. collected data in the field, conducted the analysis and led the writing of the manuscript with contributions from all other authors.

3.1. Abstract

Although mangroves support a high diversity of terrestrial and marine fauna they are being lost at high rates. One of the most important ecosystem functions provided by mangroves is their role as fish habitat. While this function has been studied in many Caribbean Islands, Indo Pacific areas and lagoon systems in the Americas there are no studies evaluating the importance of mangrove as fish habitat in lagoon systems in the Caribbean. We surveyed fish in the Ciénaga Grande de Santa Marta (CGSM), Colombian Caribbean, at six sites, five of which had different mangrove settings and one that had no mangroves. Three gillnets, parallel to the mangroves, were set at each site over six sampling cycles (n = 102); one in the mangrove, a second one further (~ 250m) from the mangrove and a third one furthest from the mangrove (~ 400m). We hypothesized that fish abundance, diversity and proportion of immature fish would be higher in mangrove habitats compared to the adjacent habitats while biomass would be higher moving away from mangroves. While the mixed effect models yielded some evidence of a habitat effect, the most important variable driving the four variables was salinity. Abundance and diversity decreased as a function of salinity while maturity slightly increased. These findings raise important conservation implications. Mangroves may be critical habitats for fish in CGSM. However, anthropogenic pressures have conditioned the salinity in the system hampering mangrove use and decreasing fish abundance. Thus, if water quality is not controlled, fish communities and fisheries are threatened despite the presence of valuable habitats.

3.2. Introduction

Mangrove habitats provide a number of different ecosystem services worldwide including provisional services such as food (Carrasquilla-Henao and Juanes 2017), wood (López-Angarita et al. 2016), supporting services (e.g. primary productivity), regulating services and cultural services (Costanza et al. 1997; Vo et al. 2012). However, mangroves are being lost at high rates (Duke et al. 2007) primarily due to anthropogenic impacts such as land-cover change, unsustainable aquaculture, pollution and overfishing (Lotze et al. 2006; Halpern et al. 2008; Spalding et al. 2010).

Juvenile and sub-adult fish use mangrove habitats as foraging areas (Green et al. 2012), shelter from predators (Laegdsgaard and Johnson 2001; Nanjo et al. 2011) or as nursery grounds, particularly in the Caribbean (Nagelkerken et al. 2000; Nagelkerken et al. 2001; Mumby et al. 2004; Chitarro et al. 2005; Nagelkerken 2009; Nagelkerken et al. 2017). In contrast, adult fish use mangrove habitats mainly as feeding ground to forage on juvenile and young of year fish in these habitats (Nagelkerken et al. 2008), but some species use them as spawning grounds (Blaber 2000).

Studies of mangrove fish habitat use in the Caribbean have mainly focused on testing the nursery hypothesis (e.g. Mumby et al. 2004), and have been conducted primarily on Caribbean islands with low mangrove area (Castellanos-Galindo and Krumme 2013), high water visibility, and where a mosaic of adjacent habitats are found; specifically seagrass beds and coral reefs (e.g. Mumby et al. 2004; Nagelkerken et al. *2017*). However, evidence for other mangrove use by different fish stages (e.g. sub-adults and adults) in the Caribbean remains scarce. Mangroves also occur on the continental

Caribbean of some countries where the geomorphological conditions differ from those present on islands. For example, on the Caribbean coast of Colombia, mangrove habitats can be found in lagoon systems where turbidity is higher due to continental freshwater discharge, salinity fluctuates as a function of precipitation and, unlike on islands, the adjacent habitat mosaic is inexistent (e.g. Ciénaga Grande de Santa Marta) or limited to one other habitat (Botero and Salzwedel 1999). In such cases, mangroves in lagoon systems in the Caribbean share some characteristics with mangroves in the Indo-West Pacific including larger mangrove coverage, turbid waters and seasonal variation in salinity. At the same time, mangrove species composition and tidal amplitude in these lagoon systems resemble those of mangroves on Caribbean islands (Spalding et al. 2010). Mangrove habitats in macrotidal (tide range > 4m) regions such as the Indo Pacific are not available throughout the full tidal cycle therefore fish only have limited temporal access to forage and/or shelter and then return to adjacent habitats such as mudflats (Ellis and Bell 2008; Igulu et al. 2014). As such, constantly inundated mangroves may also be a better food source than those with high tidal amplitudes (Nagelkerken et al. 2008).

Most studies conducted in coastal lagoon systems in the neo-tropics have described spatial distribution of fish (Rueda 2001), fish community structure (Rueda and Defeo 2003b), and fisheries aspects such as bycatch (Amezcua-Linares et al. 2009) and selectivity (Rueda 2007). However, how fish use mangroves in lagoon systems in the continental Caribbean remains unclear. In Mexico, Flores-Verdugo et al. (1990) described the fish community structure and its relationship with mangrove ecology in the Tecapán-Agua Brava lagoon system, Mexican Pacific, and Yañez-Arancibia et al. (1988)

compared fish abundance and biomass in a number of habitats in the Terminos lagoon in the Gulf of Mexico. Aburto-Oropeza et al. (2008) and Carrasquilla-Henao et al. (2013) described mangrove-fisheries relationships in several lagoon systems and a specific coastal lagoon, respectively.

Mangrove habitats in coastal lagoons have rarely been studied in Mexico in either the Gulf of Mexico or the Gulf of California and to our knowledge there have not been any mangrove habitat studies conducted in coastal lagoon systems in the Caribbean. Furthermore, while studies in Mexico have characterized the fish assemblage in lagoon systems, such previous studies (e.g. Yañez-Arancibia et al. 1988) did not attempt to disentangle the mangrove effect by consistently sampling at different distances from mangrove habitats. The purpose of this study was to determine fish ecological interactions with mangroves within a lagoon system in the Caribbean Coast of Colombia. Specifically, we asked whether fish abundance, fish biomass, fish diversity and fish maturity differ between mangrove habitats and adjacent habitats. We hypothesized that mangroves are important habitats for sub-adult fish but not for adults. Therefore we expect that abundance, diversity and number of immature fish would be highest in the mangroves and would decrease away from it while biomass would be lowest in the mangroves, since mangrove habitat provides additional structured habitat for estuarine fish and could potentially host a higher density of juveniles.

3.3. Methods

3.3.1. Study area

Ciénaga Grande de Santa Marta (CGSM) is a coastal estuarine lagoon complex system, located in Colombia's Caribbean coast and is part of the Magdalena river delta (Fig. 3.1). It is the largest of its type in the country with an approximate area of 1280km^2 and has historically been highly productive by sustaining artisanal fisheries (Botero and Salzwedel 1999; Rueda and Defeo 2003a). CGSM contains three main mangrove species, *Rhizophora mangle* (red mangrove), *Avicennia germinans* (black mangrove) and *Laguncularia racemosa* (white mangrove) that surround the system almost entirely (Fig. 3.1). Mangrove coverage was as high as 51100ha in the 1950s (Perdomo et al. 1998), but after a massive mortality event (see below) the coverage has only risen to ca. 39500ha (Ibarra et al. 2014). The system exchanges marine water through a mouth to the north, while fresh water enters the lagoon system from rivers that drain from a coastal mountain, Sierra Nevada de Santa Marta, and the Magdalena river from the east and southeast (Botero and Salzwedel 1999). Because of its importance, this ecosystem was declared a Biosphere Reserve by UNESCO and was included in the RAMSAR convention (Vilardy et al. 2011). Despite the importance of this ecosystem, it has been subject to a variety of anthropogenic disturbances that have negatively impacted the system. Connections between the lagoon and the ocean, and the lagoon and the rivers were temporarily interrupted causing hypersalinisation, which in turn, caused high mangrove and fish mortality (Perdomo et al. 1998; Botero and Salzwedel 1999). CGSM is also subject to high nutrient concentration runoff due to intensive agriculture surrounding the system, and has recently been strongly impacted by the lack of fresh water entering the system as it is being retained to irrigate crops (INVEMAR 2017). Although considerable effort has been put into restoration projects (Botero and Salzwedel 1999) the system is continuously under anthropogenic pressure.

3.3.2. Sampling

Six study sites were selected (Fig. 3.1), five of which had mangroves, Caño Grande (Cgd), Rinconada (Rin), Ciénaga La Rendonda (Clr), Aguas Negras (Agn) and Luna (Lun), and one without mangroves in the proximity, Centro (Cen). Mangrove structure has been monitored for over a decade in four of these sites (Cgd, Rin, Agn and Lun) by the Institute of Marine and Coastal Research (INVEMAR) as part of the CGSM restoration program (Botero and Salzwedel 1999). While mangroves in Clr are not sampled within the monitoring project, there is a sampling site nearby named km22. However, this site was inappropriate for fish sampling because it is located in a very narrow channel that was difficult to access and did not have sufficient space for gillnet sampling. Clr was chosen as a fishing site given its proximity to km22, which allowed us to make reasonable comparisons between the mangrove structure of Clr compared to km22.

The study site has four distinct seasons: “major dry” from December to April, “minor rainy” from May to June, “minor dry” from July to August and “major rainy” from September to November (Rueda 2001). Fish sampling occurred from May to August 2015 which falls within the minor rainy and minor dry seasons. However, due to the 2014-2016 El Niño event, the conditions were dry throughout the sampling. Cen, Cgd, Rin and Clr were sampled six times each while Agn and Lun were sampled five times each. A cycle (where all sites were sampled) lasted three consecutive days whereby two sites were sampled per day. At each site three lines were set parallel to the edge of the mangroves at different distances. Line one (Mang) was set at the edge of the mangroves, line two was set further (~ 250m) away from mangroves (Mud1) and the third line was set even further (~ 400m; Mud 2) (Table 3.1). Similarly, three parallel lines were set at different distances from a central point in Cen (as there were no mangroves here), in order to have the same number of sampling units per site. Such lines were numbered C1, C2, C3 and included in the same habitat group, mud. Thus, over the field season, 15 lines were set in Agn and Lun and 18 in the remaining four sites for a total of 102 lines (sampling units).

A line consisted of three contiguous gillnet panels of different mesh sizes, small, medium and large, to control size and species selectivity. Each panel measured about 50m in length and 1.5m in height for a total length of 150m and a fishing area of approximately 225m^2. For sites Cen, Cgd, Rin and Clr, small, medium and large mesh sizes were 5.4cm, 6.3cm and 7.6cm while for Agn and Lun mesh sizes were 5.1cm, 6.9cm and 7.6cm. The slight differences in mesh sizes were due to two different fishing crews whom, given their geographic location within the lagoon, own nets with different mesh sizes.

Lines were soaked for six continuous hours, but were checked and fish collected every two hours. At each line, dissolved oxygen (DO), salinity, temperature, pH and turbidity (Secchi disk depth) were measured. All fish samples were stored in separate bags depending on site and habitat (mang, mud1 or mud2, or C1, C2 or C3) and preserved in ice until stored and frozen in INVEMAR's facilities for later processing. Fish were identified to the lowest taxonomical level following Carpentere (2002) keys for the Western Central Atlantic. In the lab, each individual was measured (Total length, +/-0.1cm), weighed (grams, +/-0.1g) and dissected in order to determine sex and maturity when possible.

3.3.3. Data analysis

3.3.3.1. Response variables

We calculated fish abundance as the number of fish caught over a period of six hours for each line and used it as our response variable to determine the relationship with habitat and water parameters. Similarly, we calculated fish biomass per line and the total weight of all fishes caught during the six-hour period. Fish diversity was calculated based on the Shannon diversity index (Shannon 1948) because it is less sensitive to rare species than other diversity indices (Morris et al. 2014). Finally, immature and mature individuals were categorized based on gonad examination and the proportion of juveniles per line was used as the response variable.

3.3.3.2. Explanatory variables

We conducted nested (habitat within sites) linear models to determine whether environmental variables varied across sites and between habitats within sites. A model validation process was conducted for each model (Zuur et al. 2009) and if necessary the response variable (i.e. environmental variable) was log10 transformed to normalize the residuals. Variables that did not differ across sites and habitats were discarded from future analyses (i.e. pH, temperature, DO). Alternatively, if a variable differed across habitats within sites, the values where averaged by habitat (Table A3.1).

Mangrove density at each site was obtained from mangrove surveys conducted by Ibarra et al. (2014), the most updated data prior to our fish surveys. At each site mangroves were sampled in transects perpendicular to the coast using five permanent 100m^2 plots. mangrove trees were counted and identified to species. Mangrove total density was calculated by estimating the density across all plots within a site while mangrove density in the water was calculated by estimating tree density of the plots closest to the water (see Ibarra et al. (2014) for detailed explanation). Distance to mangrove for each individual line (different from those set in the mangroves) was measured in ArcGIS from the waypoint marked in the field to the edge of the mangrove where the mangrove line was set. In summary, we used two water variables (salinity and turbidity), three mangrove variables (total mangrove density, water mangrove density and distance) and habitat (where lines were set) as our explanatory variables.

3.3.3.3. Statistical models

Prior to analyzing our data all continuous explanatory variables were standardized to have a mean of 0 and standard deviation of 1 because the magnitude of the explanatory variables differed (Legendre and Legendre 2012). Fish abundance was modeled with a negative binomial distribution to account for the overdispersion in the count data which cannot be captured by the Poisson distribution (Zuur et al. 2009). Fish biomass and Shannon diversity were modeled with linear models and site was included as a random effect to account for the many sampling events within each site. To determine the relationship of the proportion of juvenile fish with the explanatory variables we conducted a Binomial Generalized Linear Mixed Model (GLMM). For these models we included site and observations (i.e. line) as random effects. The former to account for the number of sampling events within each site while the latter was to account for overdispersion (Harrison 2015).

In all cases we first constructed a full model that included all explanatory variables while a number of alternative competing models where some combinations of explanatory variables were included were used to determine the best model based on Akaike's Information Criterion (AIC). The model with the lowest AIC was chosen as the best model. However, if two or more models where within two units (i.e. $\Delta AIC < 2$) the model with the fewest parameters (i.e. most parsimonious) was chosen as the best model (Burnham and Anderson 2003). To finalize the model selection process we verified that the chosen model was a good fit by plotting the residuals against fitted values and against all covariates and assessed the patterns in the plots (Zuur and Ieno 2016). Mixed effect models where constructed and ran with the *lme4* package (Bates et al. 2015) in the statistical package R (R Core Team 2014).

3.4. Results

3.4.1. Fish abundance

A total of 1927 individuals from 34 different species and 19 families were caught throughout the sampling season. *Anchovia clupeoidea* (29.6%), *Cathorops mapale*

(19.6%), *Elops saurus* (13.1%), *Megalops atlanticus* (6.4%) and *Mugil incilis* (6.2%), were the five most abundant species across all sites and together represented ~75% of the total catches in the lagoon system. Most of the species were from estuarine environments (62%) while freshwater species represented 3.4% and marine species 8.8% of the species assemblage. However, the five most abundant species were all marine-estuarine (Table 3.2).

The best GLMM obtained by AIC_c scores retained two explanatory variables, salinity and habitat (Table A3.2). The variation explained by the random effect (site) was 31.4% suggesting fish abundance differed across sites (Fig. A3.1). Fish abundance was higher at lower salinities and decreased as salinity increased (Fig. 3.2a). In general there is a rapid (38.9%) average decrease in abundance for each salinity unit increase (-0.49, SE = 0.18, p = 0.006). However, some variability in fish abundance was observed at salinities between 34PSU and 36PSU (Fig. 3.2a). When holding salinity constant there was an observed mean fish abundance increase in habitats further from mangroves. However, the only significant difference captured by the model was between mangrove and mud2 (0.41, SE = 0.16, p = 0.009) whereby fish abundance in mud2 was 6.2% higher than in mangroves (Fig. 3.2b; Table 3.3).

3.4.2. Fish biomass

The total fish biomass collected throughout the field season was 149.29kg. Mean fish biomass from all samples (n =102) was 1.48kg ± 0.11 (mean ± SE). However, large variation was observed across lines ranging from12.2g in mud2 habitat to 6446.7g in mangrove habitat. Tarpon (*M. atlanticus*) was the species with the highest total biomass

(28.67kg) representing 19.2% of the total biomass. *E. saurus* (15.1%), *C. mapale* (13.8%), *Ariopsis sp.*(13.57%), *A. clupeoidea* (7.66%) and *M. incilis* (7.15%) together with Tarpon accounted for 63% of the total biomass across all samples.

The best model retained four explanatory variables (Table A3.3), but none were significant. Fish biomass showed a negative trend with fish abundance as salinity increased (Fig. 3.3) similar to that with fish abundance. However, unlike for abundance this relationship was only marginally significant (p=0.07, Table 3.4). Biomass in mangrove (mean = 2794.8, SE = 864.56) was ~ 30% higher than in the adjacent habitat mud1 (mean = 2176.70, SE = 749.60) (Table 3.4; Fig. A3.2), but the overall relationship between biomass and habitat characteristics was not significant (ANOVA; df = 3, F = 2.24, p = 0.09). The other two habitats, mud2 and mud (in Cen), were not significant and no clear relationship with respect to the response variable was captured by the model. Similar to the abundance model the random effect, site, explained about 20% of the variance (Fig A3.2) suggesting an important site effect.

3.4.3. Fish Diversity

Shannon diversity values ranged from 0 (multiple lines) meaning that only one species was reported for a given line to 2.11 which was obtained in mud habitat. The best model (Table A3.3) retained three covariates, salinity, turbidity and habitat category of which salinity was significant and turbidity (p = 0.07) and habitat (ANOVA; df = 3, F = 7.14, p = 0.08) marginally significant (Table 3.5).

As for abundance and biomass fish diversity decreased as a function of salinity. Thus, on average higher diversity was observed at low salinities (~ 10PSU) while low diversity was related to hypersaline (> 40PSU) environments. However, at salinities between 35PSU and 36PSU fish diversity presented some variability (Fig. 3.4a). Although, habitat was marginally significant, when controlling for salinity and turbidity mangrove habitat had higher fish diversity (1.14, SE = 0.21) than mud1 (0.86, SE = 0.21), but this was the only significant contrast (p = 0.02) (Fig. 3.4b). The relationship between fish diversity and turbidity was also negative. However, because turbidity was measured with a Secchi Disk this relationship implies that more turbid waters had higher Shannon index values. Similar to both abundance and biomass site (the random effect) captured some variability (18.8%) suggesting a site effect for fish diversity (Fig. A3.3).

3.4.4. Fish maturity

Throughout the sampling season the proportion of juvenile to adult fish across all species and habitats was close to 3:1 where juveniles represented 67% of the catch while adults only represented the remaining 33%. From the five most abundant species collected, two species (*M. atlanticus* and *E. saurus*) were exclusively juveniles while 98% of *M.incilis* were also juveniles. Conversely, *A. clupeoidea* had a more balanced proportion of juveniles and adults with 54.4% and 45.6%, respectively. When modeling the proportion of juveniles the best model retained three parameters (Table S8 in the supplement) habitat, salinity and turbidity, similar to the above models for abundance, biomass and diversity. However, unlike the previous models habitat was neither significant nor marginally significant (Table 3.6) because in all cases the proportion of juveniles was larger than adults and similar across habitats, ~75% for juveniles and ~25% for adults (Fig. 3.5).

In contrast to the negative relationship of salinity with abundance, biomass and diversity, the proportion of juvenile fish increased as salinity increased (Fig. 6a). There was a 14.5% increase in juvenile fish proportion for every unit increase in salinity (standardized scale), but no clear habitat effect. Conversely, the proportion of juvenile fish decreased at a 7% rate from turbid waters to less turbid waters, that is, from low Secchi visibility to higher Secchi visibility (Fig. 6b). The random effect site had very low variance ($3.24x10^{-10}$) while most of the random variation was accounted for by line (0.54).

3.5. Discussion

We tested the hypothesis that fish abundance, diversity and proportion of juvenile fish would decrease, while biomass would increase, as we sampled further away from mangroves. These predictions were based on the assumption that mangrove habitats are more structurally complex (Cocheret de la Morinie`re et al. 2004) and provide better feeding grounds than adjacent mudflats and thus should be more highly used by smaller fish. However, we found the opposite trend for fish abundance, whereby there was an increase in fish abundance in mud habitats (characterized as mud2 habitat) that were further away from mangroves. Our results agree with findings in Gazi Bay, Kenya where fish density and richness were lower in mangrove habitats compared with cleared habitats (Huxham et al. 2004). Similar to our study area, water visibility in Kenya was low and thus, the advantages of enhanced structure provided by mangroves as a mechanism to avoid predators could be offset by lower visibility further away from mangroves (Huxham et al. 2004). Similarly, in an estuary in Thailand, Ikejima et al. (2003) found no difference in fish density between mangrove habitat and sandy habitats. Conversely, our findings differ from others in close geographic regions in the Caribbean where fish abundance has been found to be higher in mangroves compared with adjacent habitats on an island (Nagelkerken et al. 2001). Our results also differ from a study conducted in Tanzania where fish density was found to be higher in mangroves than in adjacent habitats (Laegdsgaard and Johnson 1995). However, few studies have consistently sampled mangrove and adjacent habitats. From the limited studies to date, most have been conducted in different geomorphological and hydrodynamic settings, including estuarine and non-estuarine mangroves and regions (i.e. Americas and Indo-west Pacific). Thus, despite our geographical proximity to the study by Nagelkerken et al. (2001) their mangrove geomorphological settings differed because the mangroves found on islands are non-estuarine and not under seasonally varying hydrological conditions. Research in more stable conditions can partially explain why clear water non-estuarine mangroves on Caribbean islands have been classified as important nurseries for reef species (Barnes et al. 2012).

Although only marginally significant, we did observe a mangrove effect with respect to fish diversity, specifically between mangrove habitat and the adjacent mud habitat (mud1). Shannon diversity was higher in the mangroves compared to the mud habitat, but no trend was observed as a function of sampling away from the mangroves. In fact, the mud habitat (mud) found in site Cen had the highest Shannon diversity, but was not significantly different from other sites. The drop in diversity from mangroves to mud1 was as expected while the increase from mud1 to mud2 was not. Mangroves provide additional structural complexity that attracts more fish therefore increasing diversity while mudflats are open habitats that, while important, present more predatory risks to fish than complex habitats (Laegdsgaard and Johnson 2001). The increasing trend in diversity towards habitats further away from mangrove habitats was puzzling because habitat complexity has been shown to be a good predictor of species richness and abundance (Ferrari et al. 2016) , but could be explained by a change in the fish species composition particularly in Cgd, Rin and Cen (Fig. 3.1) where more marine conditions prevailed (see below explanation with respect to salinity and spatial variation). Our results share some similarity with results in Gazi bay where Shannon diversity was higher in mangrove creeks compared to sandy bottoms with mangroves and without mangroves nearby (Huxham et al. 2004). In both studies, diversity was higher in mangroves, but the difference between the other habitats was unclear. Interestingly, none of the mangrove variables (mangrove density and water mangrove density) were retained by our models even though we anticipated that these variables would be important predictors for our metrics. Past evidence has demonstrated that root density is a good predictor of fish density (Ronnback 1999) mainly because predation decreases and food availability increases in mangroves (Laegdsgaard and Johnson 2001). However, in Florida Faunce et al. (2004) found that prop-root density was not a good predictor of density for most fish species. In both Faunce et al. (2004) and our study, some mangrove structural variables were included in the analyses, but others were left out. Future studies should therefore include even more mangrove structural characteristics in order to increase our understanding of the importance of mangrove as habitat at local scales.

In contrast to fish abundance and diversity we did not find any habitat effects for biomass or maturity. We expected biomass to increase at habitats further from mangrove if there were larger but fewer fish and a decline in juveniles. However, the proportion of juvenile fish was high and consistent across habitats likely explaining the lack of relationship with fish biomass. If fish biomass increased away from the mangrove this would imply that a higher number of adults (larger fish) would also be present in mud habitats. Mangrove habitats on Caribbean islands have been shown to be important nurseries for reef fish based on adult fish density in adult habitats (coral reefs) as a function of distance to nursery habitats (Nagelkerken et al. 2017). While we expected to observe a similar trend within estuarine-lagoon mangroves in the same biogeographic region, our results were not consistent with these findings. A possible explanation is that the lagoon system's seascape contains less habitat variability (mangrove and mud habitat respectively) compared to Caribbean islands where mangroves, seagrass and coral reefs create a complex habitat mosaic which generates crucial connectivity among habitats for fish (Mumby et al. 2004; Nagelkerken et al. 2017). Alternatively, the whole lagoon regardless of habitat, may be acting as a nursery for many of the species observed in the study. However, this seems unlikely because some species like *Ariopsis sp*. *Eugerres plumierri* and *Cathorops mapale* are caught as adults suggesting that the lagoon is also habitat for adult individuals (INVEMAR 2017). However, these explanations must be considered with caution as the nursery function of mangrove habitats in this system remains unclear. While we were able to sample many juvenile fish the mesh size of the gillnets used (smallest mesh size was 5.1cm) did not allow us to catch fish under 10cm. Studies that have demonstrated the importance of mangroves as nurseries have used smaller mesh size gears that catch younger fish (Crona and Ronnback 2007) or by visual census techniques whereby the size of fish is determined by trained divers (Barnes et al.

2012; Nagelkerken et al. 2017) that are likely to only use these habitats until an ontogenetic shift occurs. While very small fish may not move from their nursery habitat, larger juvenile fish can move among habitats hampering our effort to determine whether mangroves are nursery habitats. There has also been debate on the proper method to sample mangrove habitat given its structural complexity (Faunce and Serafy 2006). While soaking gillnets for six hours in the edge of the mangrove allows us to capture fish that are moving in and out of the mangroves and thus permits inferences about the use of mangroves by fish (e.g. as feeding ground or shelter), conclusions about the nursery hypothesis are limited. In macrotidal mangrove systems different passive fishing gears are deployed at high tide and fish collected when the tide has ebbed (e.g. Green *et al.* 2012), but this is not possible in microtidal environments where visual censuses are frequently used. Alternatively, sampling with underwater cameras inside the prop-roots can provide valuable information on the abundance and species that use mangroves (Ellis and Bell 2013; Sheaves et al. 2016). Unfortunately, these sampling techniques are ineffective in a high turbid system like CGSM where visibility is low. Although, the lower fish size end of the spectrum is missing in our study there is evidence that many fish populations within the system are showing signs of overfishing (Narváez - Barrandica et al. 2008; INVEMAR 2017) and thus, strict fishery management strategies should be implemented in order to preserve these fish populations.

In CGSM we found salinity to be the major driving force across all metrics, but its strength was greatest for fish abundance and diversity. While also important for fish maturity and biomass, salinity showed a weaker relationship for these two metrics. Fish abundance, biomass and Shannon diversity decreased with higher salinity. However, the relationship with biomass was marginally significant. In general, most studies conducted in estuarine-mangrove habitats have found that fish abundance and metrics of fish diversity such as species richness (Sosa-López et al. 2007) and fish biomass (Lorenz 1999) are negatively related to salinity. The higher abundance and diversity at low salinities highlights the importance of brackish waters as hotspots for fish in tropical estuaries and could be categorized as ecotones (Sosa-López et al. 2007) because a number of fish follow freshwater discharges in estuarine habitats (Barletta et al. 2005). While estuarine fish can tolerate differences in salinity given their euryhaline conditions, coping with high salinities (i.e > 40PSU) generates physiological stress that many fish species avoid by migrating away from these conditions (Cowan et al. 2012).

Fish assemblages have also been found to vary with salinity (Barletta et al. 2005; Barletta and Blaber 2007; Rehage and Loftus 2007) when analyzed seasonally, that is, throughout a year of sampling where wet and dry seasons alter freshwater inflow generating salinity gradients. While a salinity seasonality has been observed in the CGSM and although our study was limited to the minor rainy and minor dry season (Rueda 2001) there was little temporal variation in salinity. In contrast, there was substantial spatial salinity variation whereby Agn had consistently mesohaline conditions (mean = 8.5PSU), Lun hypersaline conditions (mean = 51.9PSU) and marine conditions in the remaining sites (Table A3.1 and Fig. A3.1). Although we did not analyze fish assemblage composition across sites, the positive relationship between juvenile proportion and salinity can be explained by fish species composition. The most common species found in hypersaline conditions was *M. atlanticus*, the Tarpon, which has a higher salinity tolerance than other species in the estuary and was only found in the juvenile phase therefore driving maturity-salinity relationship in the opposite direction (i.e. positive). Hypersaline conditions are not only detrimental for fish but also for the mangroves themselves. While mangroves are adapted to tolerate fluctuating salinities, when these values go beyond their tolerance range mortality occurs (Barik et al. 2017). In fact, previous mangrove mortality due to hypersalinization has occurred in the CGSM (Botero and Salzwedel 1999; Vilardy et al. 2011), but salinity measurements were also used to evaluate the effect of the hydrological rehabilitation (Rivera-Monroy et al. 2011). However, we still do not clearly understand how physicochemical conditions affecting mangroves can alter the use of these habitats for fish. Nonetheless, the presence of mangrove habitats does not suffice to have healthy fish populations under poor water quality (Vidy 2000).

Three (abundance, biomass and diversity) of our four models showed considerable variation in the random effect (site) suggesting some spatial structure in the lagoon system as previously evaluated using a geostatistical approach (Rueda 2001). Sites were chosen based on mangrove characteristics (i.e. different densities), but also showed differences in salinity and water transparency. Moreover, the intrinsic location of these sites can also explain the observed variance. For example, three of the sites sampled were located in the main water body with the closest proximity to the mouth where salt water enters the lagoon. Conversely, the other three sites were located in swamp waters that had varying water physicochemical characteristics. Agn for example, is located close to the channel that feeds fresh water into the system from the Magdalena River (Fig.3.1). In contrast, Lun and Clr were located close to another channel that is blocked due to high sedimentation and has historically lacked maintenance (Vilardy et al. 2011). Based on our results, the main driver of the spatial variation is also salinity which is consistent with other lagoon systems in the Americas (Sosa-López et al. 2007). However, in sites with similar salinity there was also considerable variation in abundance, diversity and biomass suggesting that site characteristics may play a synergistic effect on these metrics. Future studies should be designed explicitly to disentangle these effects.

While mangrove habitats had small effects on some of the common metrics used to classify them as nursery areas, in general we were unable to observe a clear nursery function in mangroves in the CGSM mostly due to our sampling design. However, we do not discard the nursery function provided by mangrove habitats in this system. Rather, we suggest targeting smaller fish sizes in the mangroves in order to determine the nursery capacity of mangroves en continental Caribbean estuarine systems. However it is possible that, in heavily impacted ecosystems where there are many anthropogenic stressors, this function may be altered or reduced. Therefore, testing the nursery hypothesis in lagoon systems that resemble characteristics of both non-estuarine mangroves in the Caribbean (e.g. low tidal regimes and clear waters) and Indo-West Pacific characteristics (high turbidity and large estuaries) should be compared to less degraded systems. We consider that urgent conservation strategies are needed in the CGSM in order to re-establish the normal hydrodynamic regime as an attempt to preserve fish populations and mangrove ecosystems. The CGSM is the largest lagoon system in Colombia and was once highly productive in terms of fish biomass providing subsistence fisheries to many locals who make their living exclusively from the system (Rueda et al. 2011).

In conclusion, mangrove habitats in CGSM are used by fish indistinguishably from adjacent habitats, however there is some evidence that mangroves can hold a higher species diversity than muddy habitats. Instead, salinity was an important driver for fish abundance, diversity and maturity. Although the CGSM has fluctuating hydrological regimes based on seasonality, some salinity values observed were higher than previously observed in the system. As a result, these high values coupled with other anthropogenic activities may be hampering our ability to observe the benefits of mangroves as critical habitats for fish. Thus, we suggest implementing conservation strategies as soon as possible to mitigate the impacts that the system is suffering.

Table 3.1. Table showing the mean distance to mangroves of gillnets set in different habitats at each site for all cycles combined. Mud 1 refers to the habitat of the gillnet set further from mangroves and Mud 2 is the habitat of the gillnet set farthest from mangroves. Gillnets set at the mangroves were omitted because the mean distance and corresponding standard deviation are 0.

Site	*Habitat*	*Mean distance to mangrove (m)*	*Standard deviation*
Caño Grande (Cgd)	Mud 1	236.42	57.24
	Mud 2	448.39	110.16
Rinconada (Rin)	Mud 1	253.64	26.45
	Mud 2	462.83	31.86
Ciénaga La Redonda (Clr)	Mud 1	133.01	25.74
	Mud 2	366.25	54.49
Aguas Negras (Agn)	Mud 1	320.43	122.41
	Mud 2	585.40	138.37
Luna (Lun)	Mud 1	236.25	19.67
	Mud 2	381.07	53.15

Table 3.2. Total fish abundance and relative abundance by species collected across all sites during the sampling season in Ciénaga Grande de Santa Marta, Colombian Caribbean. In column environment fish categories are as follows ME = Marine-estuarine, M = Marine and F = Freshwater.

Species	***Family***	***Environment***	***Total Abundance***	***Relative abundance (%)***
Anchovia clupeoidea	Engraulidae	ME	570	29.6
Cathorops mapale	Ariidaae	ME	377	19.6
Elops saurus	Elopidae	ME	253	13.1
Megalops atlanticus	Megalopidae	ME	123	6.4
Mugil incilis	Mugilidae	ME	120	6.2
Micropogonias furnieri	Sciaenidae	ME	104	5.4
Ariopsis sp.	Ariidaae	ME	99	5.1
Eugerres plumieri	Gerreidae	ME	56	2.9
Strongylura marina	Belonidae	ME	36	1.9
Oligoplites palometa	Carangidae	ME	33	1.7
Diapterus rombeus	Gerreidae	ME	25	1.3
Bairdiella ronchus	Sciaenidae	ME	22	1.1
Mugil liza	Mugilidae	ME	13	0.7
Cetengraulis edentulus	Engraulidae	ME	12	0.6
Caranx hippos	Carangidae	M	12	0.6

Achirus lineatus	Achiridae	ME	11	0.6
Stellifer venezuelae	Sciaenidae	ME	10	0.5
Trachelyopterus insignis	Auchenipteridae	F	8	0.4
Leporinus muyscorum	Anostomidae	F	7	0.4
Ctenolucius hujeta	Ctenoluciidae	F	5	0.3
Oligoplites saurus	Carangidae	ME	5	0.3
Gerres cinereus	Gerreidae	ME	5	0.3
Diapterus auratus	Gerreidae	ME	4	0.2
Centropomus undecimalis	Sciaenidae	ME	4	0.2
Mugil curema	Mugilidae	ME	2	0.1
Caquetaia kraussii	Cichlidae	F	2	0.1
Oreochromis niloticus	Cichlidae	F	2	0.1
Astyanax fasciatus	Characidae	F	1	0.1
Curimata mivartii	Curimatidae	F	1	0.1
Prochilodus magdalenae	Prochilodontidae	F	1	0.1
Chloroscombrus chrysurus	Carangidae	M	1	0.1
Oreochromis spp.	Cichlidae	F	1	0.1
Lutjanus sp.	Lutjanidae	M	1	0.1
Pimelodus blochii	Pimelodidae	F	1	0.1

Table 3.3. Parameter estimates for the best GLMM based on AIC_c scores for fish abundance relationships with site as a random effect. Mud 1 = habitat further from mangroves; Mud 2 = habitat furthest from mangrove and Mud = mud habitat in site Cen.

Coefficient	Estimate	Standard error	P value
Intercept	2.3	0.27	$< 2.2^{-16}$
Salinity	-0.49	0.18	0.006
Mud 1	0.07	0.16	0.67
Mud 2	0.41	0.16	0.009
Mud	1.08	0.63	0.09

Table 3.4. Parameter estimates for the best linear mixed model based on AIC_c scores for fish biomass relationships with site as a random effect. Mud 1 = habitat further from mangroves; Mud 2 = habitat furthest from mangrove and Mud = mud habitat in site Cen.

Coefficient	Estimate	Standard error	P value
Intercept	2794.80	864.56	0.0018
Salinity	-393.20	182.52	0.07
Turbidity	-185.41	12.18	0.13
Distance	2507	1525.09	0.10
Mud1	-618-09	256.07	0.02
Mud2	-364.64	322.08	0.26
Mud	-5817.82	4110.19	0.16

Table 3.5. Parameter estimates for the best linear mixed model based on AIC_c scores for fish diversity (Shannon Diversity) relationships with site as a random effect. Mud 1 = habitat further from mangroves; Mud 2 = habitat furthest from mangrove and Mud = mud habitat in site Cen.

Coefficient	Estimate	Standard error	P value
Intercept	1.14	0.21	0.0039
Salinity	-0.260	0.12	0.04
Turbidity	-0.11	0.06	0.07
Mud1	-0.27	0.11	0.01
Mud2	-0.04	0.11	0.74
Mud	0.74	0.49	0.21

Table 3.6. Parameter estimates for the best generalized linear mixed model (family binomial) based on AIC$_c$ scores for fish maturity relationships with site and observation (i.e. line) as a random effect. Mud 1 = habitat further from mangroves; Mud 2 = habitat furthest from mangrove and Mud = mud habitat in site Cen.

Coefficient	Estimate	Standard error	P value
Intercept	1.36	0.23	5.12x10^{-9}
Salinity	0.58	0.13	4.78x10^{-6}
Turbidity	-0.28	0.14	0.04
Mud1	0.02	0.31	0.94
Mud2	-0.16	0.3	0.58
Mud	-0.5	0.33	0.14

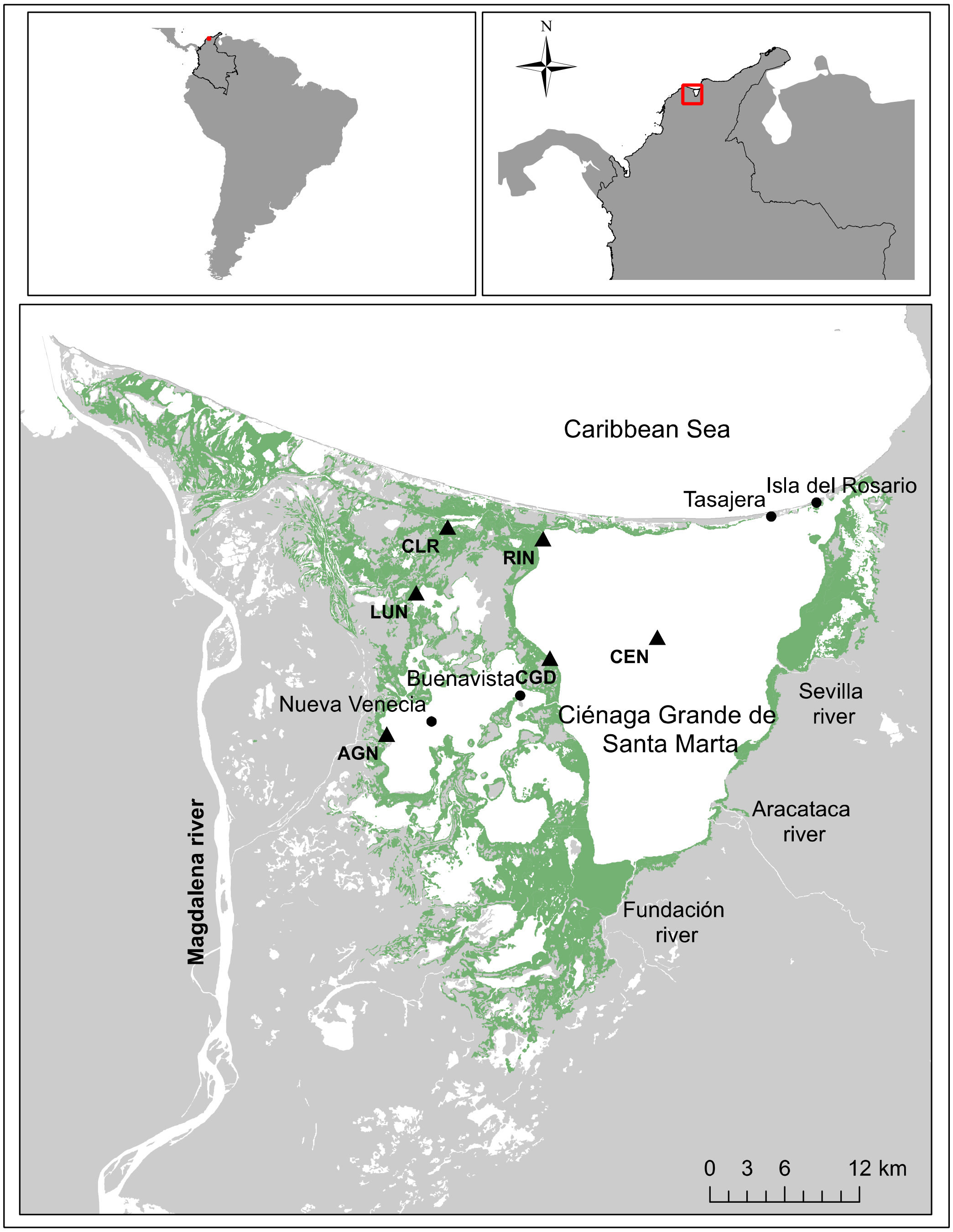

Figure 3.1. Map of Ciénaga Grande de Santa Marta (CGSM) showing the six sites (▲) where sampling took place. CEN = Centro; CGD = Caño Grande; RIN = Rinconada; AGN = Aguas negras; LUN = Luna; and CLR = Ciénaga La redonda.

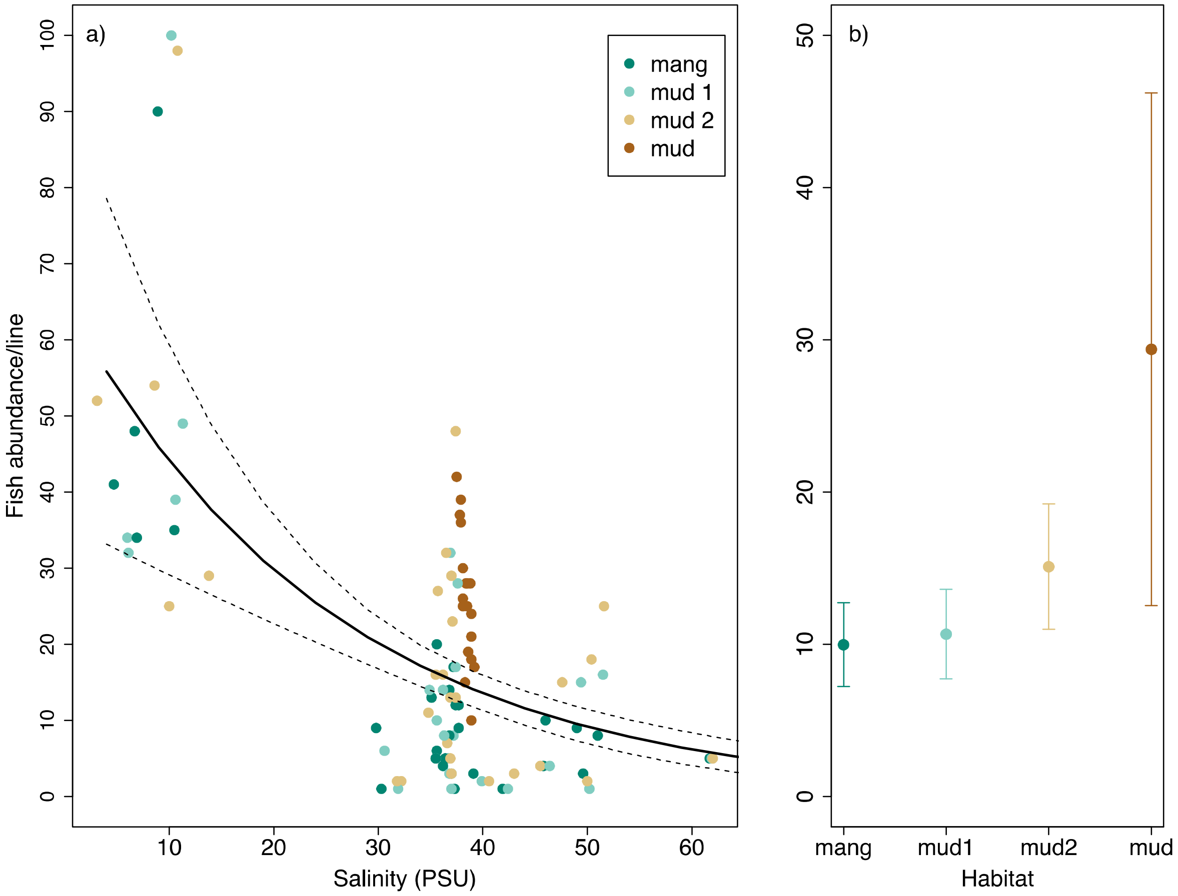

Figure 3.2. Fish abundance estimate for (a) salinity and (b) habitats from the best model based on AICc scores. In panel (a) black line represents model fit and dotted lines represent 95% confidence intervals. In panel (b) points represent predicted means from the mixed effects model while error bars represent estimated standard errors. Habitat categories are as follows: mang = mangrove habitat; mud1 = mud habitat further from mangroves; mud2 = mud habitat furthest from mangroves and mud = habitat in site Cen.

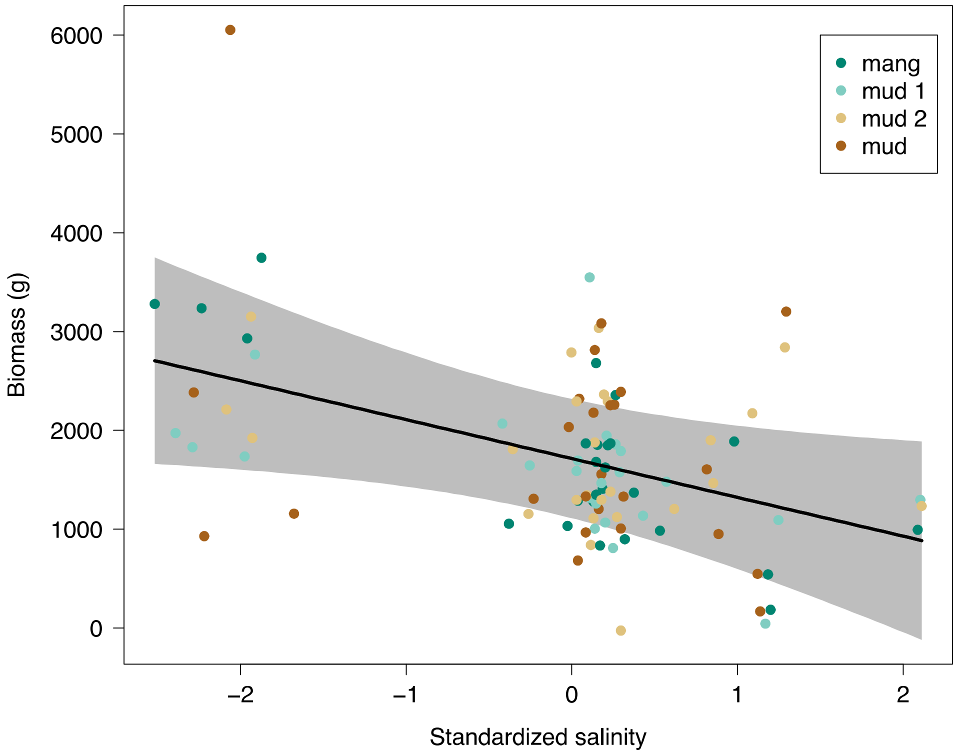

Figure 3.3. Fish biomass estimates for salinity from the best model as evaluated by AICc scores. Black line represents the fitted model while the grey band represents 95% confidence intervals. Habitat categories are as follows: mang = mangrove habitat; mud1 = mud habitat further from mangroves; mud2 = mud habitat furthest from mangroves and mud = habitat in site Cen.

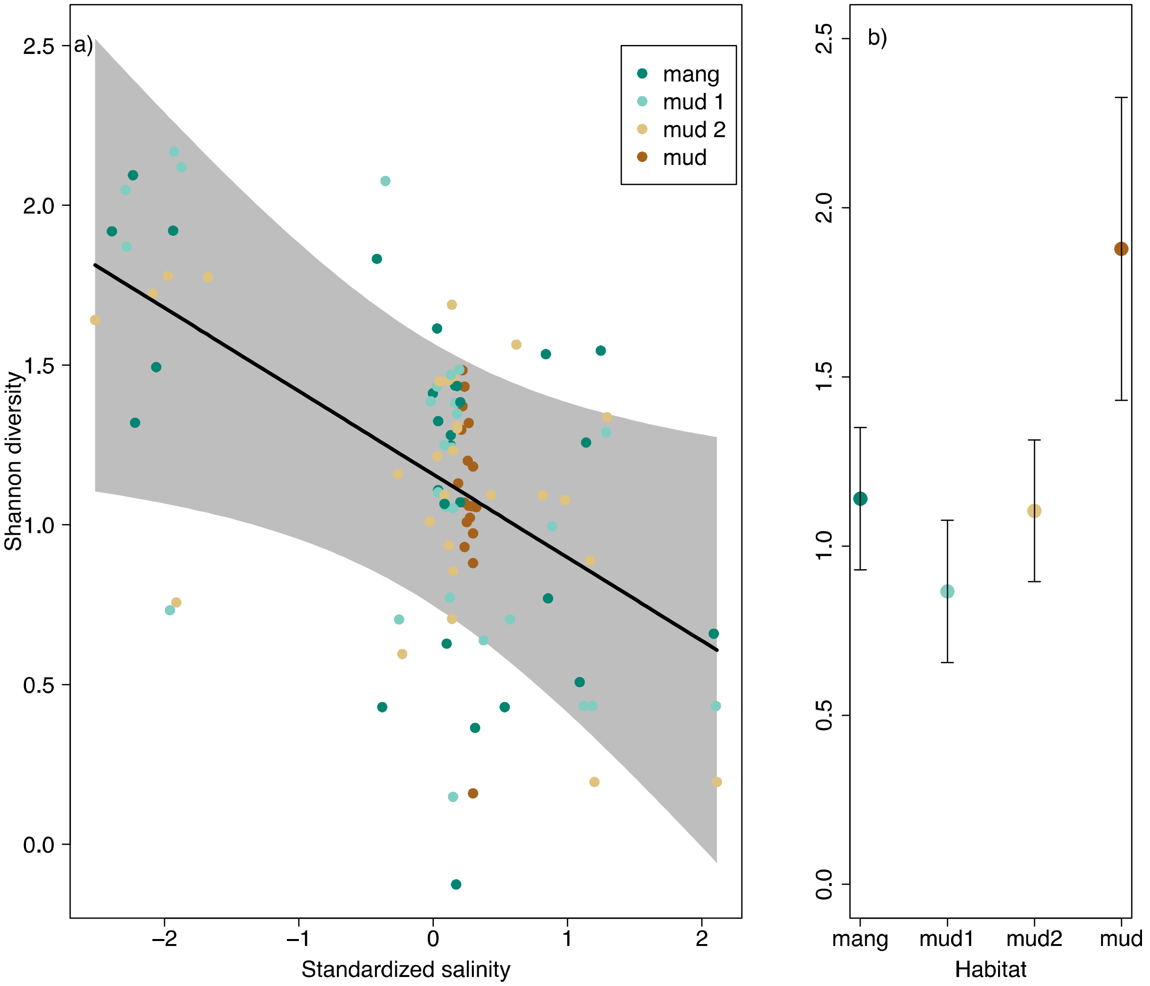

Figure 3.4. Fish diversity estimate for (a) salinity and (b) habitats from the best model based on AICc scores. In panel (a) black line represents model fit and shaded area 95% confidence intervals. In panel (b) points represent predicted means from the mixed effects model while error bars represent estimated standard errors. Habitat categories are as follows: mang = mangrove habitat; mud1 = mud habitat further from mangroves; mud2 = mud habitat furthest from mangroves and mud = habitat in site Cen.

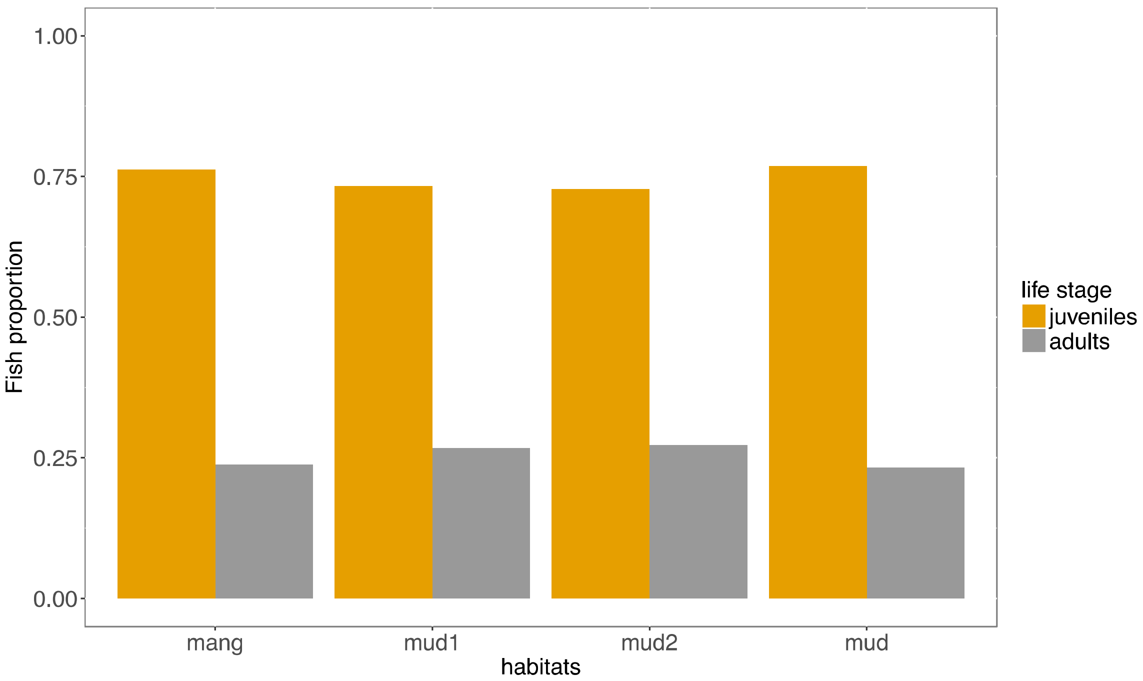

Figure 3.5. Proportion of juvenile and adult fish across habitats. Habitat categories are as follows: mang = mangrove habitat; mud1 = mud habitat further from mangroves; mud2 = mud habitat furthest from mangroves and mud = habitat in site Cen.

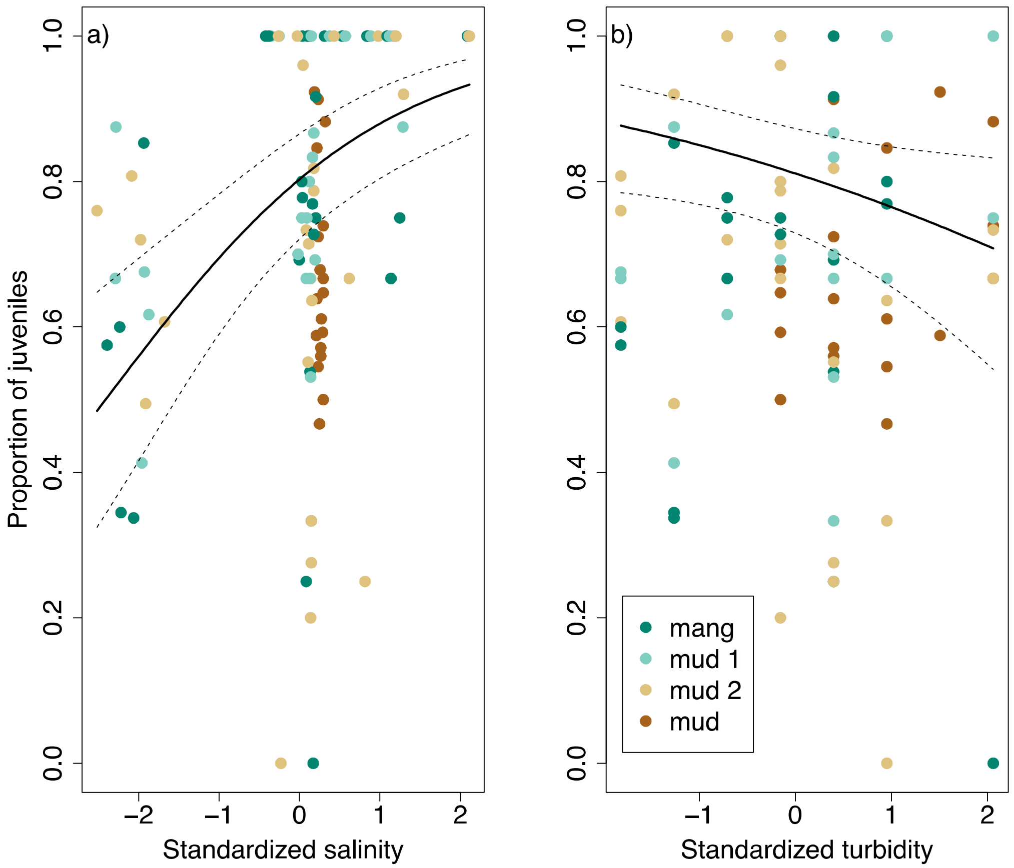

Figure 3.6. Proportion of juvenile fish against (a) salinity and (b) turbidity for the best model based on AICc scores. Thick black lines represent model fit and dotted lines represent 95% confidence intervals. Note that lower values of turbidity mean more turbid waters as the measurements were conducted with a Secchi Disk. Habitat categories are as follows: mang = mangrove habitat; mud1 = mud habitat further from mangroves; mud2 = mud habitat furthest from mangroves and mud = habitat in site Cen.

Chapter 4 - The mangrove-fishery relationship: A Local Ecological Knowledge perspective

4.1. Abstract

Mangroves are one of the major ecosystems in coastal areas of tropical and subtropical regions, are critical habitats for fish and crustaceans, and provide a number of different ecosystem services to people who live in proximity to them. Local Ecological Knowledge (LEK) transmits the knowledge gained from a local system due to intimate relationships with the environment through generations. While mangrove uses have been widely documented based on LEK studies, seldom has this approach been used to analyze the mangrove-fishery relationship. By conducting semi-structured interviews (n = 82) with fishers in three different villages surrounding the most important lagoon system in the Colombian Caribbean, we evaluated the fishing and gear spatial distribution, mangrove use and mangrove-fishery linkage. Fishing is concentrated close to fishers' villages though there is some gear and species spatial variation across villages. Mangroves are mainly used for firewood, construction and for fishing gear, but uses vary across villages. Respondents believe that mangroves are critical habitats for fishery resources and consider that the resource would be in jeopardy in the absence of mangroves. Given that the system is highly degraded and conservation and fishery management plans are urgently required, we suggest that combining scientific knowledge with LEK in the planning and implementation of restoration and conservation plans will increase the chances of such programs being successful.

4.2. Introduction

Marine coastal ecosystems are considered critical in terms of the number of ecosystem services they provide (Costanza et al. 1997). However, they have suffered considerable degradation (Lotze et al. 2006; Halpern et al. 2008) such as over fishing (Jackson et al. 2001), habitat loss, pollution and climate change (Crain et al. 2009). This degradation is exacerbated by a steady human population growth in coastal habitats (Vitousek et al. 1997; Neumann et al. 2015).

In the tropics, many people rely on the resources provided by mangrove-dominated estuaries, such as fish production (Blaber 2000) whereby catches are directly related to mangrove area (Carrasquilla-Henao and Juanes 2017), wood products (Dahdouh-Guebas et al. 2006) for subsistence and protection against natural disasters (Danielsen et al. 2005). Although fishing is the major economic activity conducted by fishers to sustain their families (Walters et al. 2008), many other natural resources are exploited by locals, particularly wood-derived products from mangroves (e.g. Dahdouh-Guebas et al. 2006; Palacios and Cantera 2017). However, these other activities are not conducted as intensely as fishing, yet it is important to satisfy critical needs such as construction materials and fuel (Walters et al. 2008). While harvesting mangroves by locals may not be as detrimental as many other threats mangroves face (Valiela et al. 2001; López-Angarita et al. 2016), selective harvesting can considerably change forests structure and ecological processes (López-Hoffman et al. 2006; Palacios and Cantera 2017). These ecological changes can have consequences for fish production because mangroves have been shown to be critical habitats for fish and prawns. In many cases they are good nursery habitats (Nagelkerken et al. 2008; Nagelkerken et al. 2017), serve as feeding grounds, decrease predation risk for many fish species (Laegdsgaard and Johnson 2001) and can enhance fish abundance of nearby coral reefs (Serafy et al. 2015).

Local Ecological Knowledge (LEK) can be an important source of information for conservation and management. LEK is the oral transmission of knowledge through generations about the relationship that humans have with the environment. As such, it is cumulative in nature (Berkes et al. 2000). Over the course of their lives, fishers spend many hours a day interacting intimately with the environment and therefore, acquire valuable information about collecting resources efficiently, weather patterns, species' life histories observations and predator-prey relationships. In rapidly changing ecosystems, where long-term experiments are scarce, LEK is extremely helpful in establishing conservation plans when integrated with scientific data and where no scientific data exists (Drew 2005; Brook and McLachlan 2008; Ban et al. 2013). Given that in the tropics fishers exploit multiple resources from the same area, they have vast knowledge about local ecological patterns. However, to date, studies including LEK in mangrove-dominated systems have focused primarily on mangrove use (Hernández-Cornejo et al. 2005; Dahdouh-Guebas et al. 2006; Hussain and Badola 2010; Palacios and Cantera 2017) and lack detailed explanations of fishing activities, fishing gear or the spatio-temporal distribution of fishing within these systems, despite small-scale fisheries being the predominant activity in tropical and subtropical mangrove dominated estuaries. LEK has been effective for establishing Marine Protected Areas (Scholz et al. 2004; Aswani and Lauer 2006; Mellado et al. 2014), regulating fishery catches and determining spatio-temporal distribution of catches (Hall and Close 2007; Moreno-Báez et al. 2010; Yates and Schoeman 2013). Furthermore, the majority of attempts to explain the importance of mangroves for fishery resources have failed to include the major stakeholders' opinions on how mangroves help support local fisheries. An exception is a study in a lagoon system in the Sea of Cortez in Mexico, that mapped the locations where fishers were engaged in fishing and estimated the mangrove distance at which the activity was carried out (Carrasquilla-Henao et al. 2013). Interviewing fishers about their perceptions on the importance of mangroves for fisheries from a LEK perspective is a novel complement to the myriad of studies that have contributed to the understanding of mangrove-fish(eries) relationships based on scientific knowledge.

The purpose of this study was to use fishers' LEK to understand their perceptions of the mangrove-fishery relationship in the Ciénaga Grande de Santa Marta (CGSM), a continental lagoon system located in the Caribbean coast of Colombia. We conducted semi-structured interviews with fishers who had fished for more than 10 years and asked three major questions a) what is the fishing spatial distribution in the CGSM? b) what are the major reasons to harvest mangrove? and c) what is the fisher's perception of the mangrove-fishery linkage in the CGSM?

4.3. Methods

4.3.1. Study site

The CGSM is the largest lagoon estuarine system in the continental Colombian Caribbean with an approximate area of 1280km^2 that includes a number of different water bodies such as swamps, channels and marine waters (Fig. 4.1). The lagoon system also supports a large mangrove forest that provides habitat for a high diversity of taxonomic groups belonging to both marine and terrestrial species (Vilardy et al. 2011). Thus, the CGSM is a highly productive system that contributes a large percentage of the fish catch in the Colombian Caribbean (Rueda et al. 2011). The estuary's hydrodynamics are controlled by a connection with the ocean in the northernmost part of the lagoon while freshwater is provided by rivers that drain from the Sierra Nevada de Santa Marta mountain range and from Colombia's main river, the Magdalena River (Botero and Salzwedel 1999; Vilardy et al. 2011). Despite its importance, the system has been under constant anthropogenic pressure for over five decades for a number of reasons. Firstly, the sea-lagoon system connection was interrupted when a highway was constructed along the coastline in the 1950s. The freshwater flow to the system was also interrupted in the 1990s in order to prevent flooding in growing agricultural areas surrounding the CGSM (Botero and Salzwedel 1999; Vilardy et al. 2011). As a consequence, the system experienced abnormally high salinity concentrations, eliciting a massive mangrove and fish mortality. While a large restoration project was established to recover the freshwater and marine inputs to the system (Botero and Salzwedel 1999), currently the CGSM is still under constant threat due to multiple stressors such as pollution, sedimentation preventing fresh water flow and overfishing (Narváez - Barrandica et al. 2008).

There are eleven human communities surrounding the study area; most of their population relies on the natural resources from the CGSM such as fisheries and mangrove products. Most people in these communities live in extreme poverty such that many of their basic needs are unsatisfied and most people have little or no education (Rueda et al.

2011; Vilardy et al. 2011). However, villages located close to large cities (i.e. Ciénaga) or close to the highway connecting two large cities (i.e., Barranquilla and Santa Marta) have better conditions as they can market their catches more easily (Rueda et al. 2011).

4.3.2. Semi-structured Interviews

We conducted semi-structured interviews with fishers who had fished for at least 10 years in the lagoon system from three different fishing villages, Tasajera, Isla Rosario and Nueva Venecia. Tasajera and Isla Rosario are located next to the highway that connects both major cities; Nueva Venecia is a stilt village (Fig. 4.1). Theses sites were chosen because fishers have worked closely with INVEMAR over the years. Interviews lasted between 45 – 60 minutes and were conducted by convenience sampling mainly at the landing sites. Field assistants, the same people who conduct fishery surveys for INVEMAR, conducted most of the interviews. Interviewers were trained and provided with an instructional document on how to conduct and record the interview. The lead author conducted interviews with each assistant so they understood the procedure until they felt confident about conducting interviews by themselves. While LEK approaches may be biased due to interviewee dishonesty (Close and Hall 2006), we consider that such bias is decreased when fishers trust a familiar interviewer. A total of 82 interviews were conducted over a two-month period to fishers who had fished for at least 10 years in the lagoon system (from June to August 2015): 19 in Nueva Venecia (NV), 39 in Isla Rosario (IR) and 24 in Tasajera (TA).

The interviews consisted of two parts. First, fishers were asked about their fishing activity and their perceptions on the importance of mangroves for their activity (See appendix A4 for complete interview). In this section we asked them to identify the most important species caught on different temporal scales (i.e., at present (2015), 5 years ago (2010) and 10 years ago (2005)), what fishing gear they used to catch the species and their economic dependence on fishing activity. We also queried them about their perception of both catch and size of species caught in 2010 and 2005 relative to 2015. With respect to mangroves, we aimed to gather information about how fishers use mangroves in their daily activities and their opinion on the mangrove-fishery linkage, more specifically, if they considered that mangroves were important for their catches.

The second part of the interview required the fishers to draw a polygon of their preferred fishing areas at the same temporal scales as above. This allowed us to relate the spatial location of their catch with the species caught and the fishing gear used to create an attribute table for each polygon. Each fisher was presented with a 4 x 4 gridded map (Fig. A4.1) of the whole study area and was asked to show where their fishing area was on the map. After they had pointed to a specific grid (*e.g.* A4) the interviewer went to the zoomed-in, single page grid and then asked the person to draw their preferred fishing area. This procedure was repeated three times, one for “at present” (2015), one for “5 years ago” (2010) and one for “10 years ago” (2005). The polygons were digitized as a single shape file in ArcGIS 10.4 with the Georeference tool using the gridded map described above as reference. A unique identifier that included the village where the interview was done and the temporal scale was entered in the attribute table to relate the spatial file with the fishing information.

4.3.3. Data analysis

Contingency tables were created with the number of responses per fishing village and the data were analyzed with log-linear models in order to determine whether responses differed across fishing villages and time periods. Log-linear models were first fitted with a saturated model (i.e. no degrees of freedom) and then the highest interaction was removed. To determine if the interactions were significant, partial Chi-square tests between the two models were carried out (Crawley 2013). If the interactions were not significant the conclusion was that the response for any given category was independent. A significant interaction suggested that the answers were not independent for each village and therefore, there was an association between the variables measured. All statistical analysis were conducted in R (R Core Team 2016).

To determine the spatial distribution of the main fishing areas, fishing gear used and most important species caught in each fishing village and all villages combined, we counted the number of overlapping polygons for each village at a given time. For example, if two different fishers from the same village drew a polygon, and these overlapped, the count would be two. Next, we created a raster dataset from the output shape file with a cell size of 100m. This resolution was chosen to allow for potential biases in the drawn polygons and the smoothing technique chosen (Turner et al. 2015). Thus, high count values (i.e. high polygon overlap) represented a higher area used while lower values represented areas used by fewer fishers. Finally we applied a filter in order to smooth the raster dataset and have a cleaner representation of the fishing area.

4.4. Results

4.4.1. Fisheries

Respondents had fished in the lagoon system for an average of 33.7 years, ranging from 10 years to 55 years. The time in years that fishers from IR had fished (mean = 39.82 years ± SD = 10.99 years) was ~10 years greater than in TA (28.04 years ± SD = 10.63 years) or NV (28.58years ± SD =10.56 years). All of the income from respondents from TA and NV depended on their catch while, for 84.6% of the fishers in IR, all their income depended on fishing. However, 15.4% of the fishers in IR had additional income despite most of their income relying on their daily catch.

Overall, the most common fishing areas were highest in the northern part of the CGSM although a large area used by fewer fishers was observed in the southwestern area of the CGSM (Fig. A4.2). The most common fishing grounds of the three villages were close to their landing site suggesting short travel distances. However, in 2005 some fishers from IR were fishing in the southern part of the CGSM. Similarly, the areas used by fishers in NV were more widespread and further from their village in 2005 compared to 2015 (Fig. 4.2).

In general, shrimp nets were the most important fishing gear used followed by cast nets and crab traps. Shrimp nets were most commonly used in both IR (41.03%) and TA (37.5%), while cast nets were more commonly used in NV (42.11%). Although crab traps and gillnets were the only two gear types shared across the three villages, the use frequency differed across the communities (Fig. A4.3) as we found no association between fishing gear and villages ($\chi^2 = 6.59$, df = 5, p = 0.25). However, interesting spatial patterns emerged related to the different gear used. For example, shrimp nets, encircling gillnets and gillnets were mainly used in the north. Shrimp nets were used along the whole northern border while encircling gillnets were limited to one specific area. Conversely, seine nets were mainly limited to the southwestern area of the lagoon system and their use was more widely distributed than other gears. Similarly, cast nets covered a large area although they were mostly used at the mouth of the lagoon (Fig. 4.3).

Shrimp, crabs (*Callinectes* sp.) and striped mojarra (*Eugerres plumieri*) were the most caught species across all villages and throughout the three time-points (Fig A4.4). However, the species caught did not differ in the three time periods across villages ($\chi^2 = 13.48$, df = 30, p = 0.9). Nonetheless, when removing the three-way interaction from the model we observed that villages select for different species ($\chi^2 = 211.5$, df = 26, $p < 0.001$). For example, shrimp, crabs and striped mojarra were the most important species caught in IR whereas shrimp, ladyfish (*Elops saurus*) and striped mojarra were the most important species in TA. Conversely, tilapia (*Orechromis niloticus*), crabs and a mullet (*Mugil incilis*) were the most targeted species in NV. Similarly, catch diversity (i.e. number of species) was highest in IR (9 species) followed by TA (8 species) and NV (5 species). Although some variation was observed across time periods, there were no statistical differences ($\chi^2 = 7.76$, df = 15, p = 0.9) suggesting that fishers have selected the same species over time (Fig. 4.4).

Shrimp and crab distribution within the lagoon system resemble the two fishing methods that target such groups. The spatial patterns for fish species are a function of the village and the gear used. Striped mojarra, for example, is highly caught where fishers drew polygons for encircling gillnets, but were also caught in the mouth of the lagoon system. However, striped mojarra had a small area relative to the whole system represented. In contrast, *M. incilis* was caught over a wider area across the system although it was not as highly targeted as *E. plumieri*. *O. niloticus*, which was only targeted by fishermen in NV showed the most southern distribution and had some area overlap with *M. incilis* and *Ariopsis sp.* (Fig. 4.5).

The three-way interaction for fishers' perception on whether the catch has increased, stayed the same or decreased (catch X village X time) was not significant ($\chi^2 = 7.76$, df = 4, p = 0.1), but both catch X village ($\chi^2 = 39$, df = 4, p < 0.01) and catch X time ($\chi^2 = 19.83$, df = 2, p < 0.01) were significant. Thus, the catch perception differs across villages, particularly in 2010 where most fishers (66.6%) from TA considered catches to be the same as in 2015. However, fishers from all three villages agreed that catches were worse in 2015 compared to 2005 (Fig. 4.6a and Fig. 4.6b).

We also asked fishers to determine whether fish/crab size was smaller or larger in 2015 relative to 2010 and 2005. Size perception of their catch differed across villages ($\chi^2 = 12.25$, df = 4, p = 0.01); fishers in TA (62.5%) and NV (52.63%) considered that size of individuals caught in 2010 compared to 2015 were similar, while respondents in IR considered that individuals were smaller (56.41%) in a higher proportion compared to the other two villages (Fig. 4.6c). Fisher's size perception also differed over time ($\chi^2 = 4.29$, df = 1, p=0.038). While fishers considered that the size was similar and in some cases larger in 2010, the perception in 2005 changed to a majority of respondents in the three villages considering that individuals were either smaller or similar in size compared to 2015 (Fig. 4.6d).

4.4.2. Mangroves

In the CGSM, fishers from the three villages used mangroves for different purposes ($\chi^2 = 70.10$, df = 10, $p < 0.01$). Mangrove as firewood was used by all fishers in TA and by 87.18% of fishers in IR, but minimally used in NV (26.32%). However, the use of mangroves to construct houses and poles (herein defined as a tool used to move a canoe) is greater, and unique respectively, in NV compared to IR and TA. Fishing gear derived from mangroves was higher in IR (56.41%) than in TA (4.17%) and NV (26.32%). Less frequently, mangroves are also used for canoe construction and for work tools among the interviewed fishers in the three villages (Fig. 4.7a).

All fishers agreed that there is a mangrove-fishery relationship in CGSM. However, people from different villages differed in their opinion on how fishery resources use mangroves ($\chi^2 = 26.73$, df = 10, p = 0.029). The most uniform responses across villages were that organisms use mangroves as feeding areas and habitat in general. A high number of interviewees from TA (57.14%) and IR (39.28%) compared to NV (3.57%) considered that mangroves are important nursery areas for the species they catch. The importance of mangroves as refuge was also mentioned by fishers in all three villages, but was more prevalent in IR (55%) and NV (35%). The use of mangroves as breeding areas was only mentioned by fishers in IR (83.3%) and NV (16.6%), while the use of mangroves as spawning areas was only mentioned in IR (Fig. 4.7b).

When fishers were asked what would happen to fishery resources in the absence of mangroves, all of their responses suggested a negative impact on the resource, although the reasons differed across villages ($\chi^2 = 78.73$, df = 18, $p < 0.001$). Fishers in IR showed the most diverse suite of responses ranging from death from a given cause (e.g., warmer water) to lack of reproduction. Respondents from TA and NV provided less descriptive answers and mostly stated that fish would die or disappear (Table 4.1).

4.5. Discussion

Our findings highlight the importance of including LEK as an additional and critical source of information to include in potential management plans (Ban et al. 2009; Ban et al. 2013). Our study is the first to conduct an in-depth analysis of fishing activity, mangrove use and the mangrove-fishery relationship, by interviewing major stakeholders in the system to understand the relationship from a holistic perspective. In general, we found that fishers don’t travel long distances to their preferred fishing ground, fishing occurs close to mangroves, fishing areas have not changed considerably over the three time periods, and that there are spatial differences in fishing grounds based on fishing gears. Our results also show that fishers harvest mangrove trees for a number of products and that these vary according to fishing village. Lastly, with a LEK approach we found overwhelming evidence of the importance of mangroves for their fishing activity. All fishers agreed that there is a linkage between mangroves and fishers, although the perceived importance of mangrove for fishing resources differed among fishers.

4.5.1. Fishing activity

The most commonly used fishing grounds (i.e. where more polygons overlapped) were at short distances from fishers' home villages which suggests short travel distances for their activity. Similar patterns have been observed in a lobster fishery in the UK (Turner et al. 2015) and in the Patos lagoon system in Brazil (De Freitas and Tagliani 2009). Several explanations have been proposed. For example, there seems to be territoriality across villages, which elicits the spatial differentiation (St. Martin and Hall-Arber 2008; Turner et al. 2015). While this is a plausible explanation in the CGSM, particularly for Isla Rosario (IR) and Tasajera (TA) which are at close distance to each other (~ 3km), we found some overlap in fishing grounds for these two villages. Consequently, we consider that the mechanism driving the observed spatial distribution is mainly based on social-economical constraints which are pervasive in mangrove dominated systems in developing countries (De Freitas and Tagliani 2009). Because these communities live in extreme poverty they need to maximize their income (Rueda et al. 2011; Vilardy et al. 2011). As a result, traveling shorter distances implies less fuel consumption. Moreover, when traveling further distances fishers in the CGSM use sailing techniques in order to save fuel or because they do not have a power motor (personal observation). These economical limitations have been shown to limit catch trading due to difficult transport to markets and vessel size constraints impeding resource exploitation at larger distances from the coast (Metcalfe et al. 2016). We also found little spatial change over the three time periods consistent with the lobster fishery in the UK (Turner et al. 2015). The pattern in Nueva Venecia suggests that fishers were fishing further away from the village earlier in time. Similarly, there are some marked areas in the southern part of the lagoon system for IR. Although more resources could have been available in the past, which merited traveling longer distances, an alternative explanation is that some fishers used to live in a different village and have moved to the village where they were interviewed. Indeed, Colombia's armed conflict has induced fisher displacements to villages closer to the large cities (Vilardy and Renan-Rodriguez 2011). There was also spatial variability across gear types in the CGSM that can be related to territoriality or habitat use by target species (St. Martin and Hall-Arber 2008). For example, shrimp nets are used along the northern area of the CGSM close to the mouth of the sand spit that separates the lagoon system from the ocean. In contrast, gillnetting and crab trap use are more limited in space while the mouth of the system is a priority area for fishers with cast nets. This northern area has been previously identified as an important fishing ground but that analysis was not spatially plotted (Torres-Guevara et al. 2016). Fishing areas by village, gear type and over time are, for the most part, close to mangrove areas in the CGSM. The spatial visualization of this pattern is consistent with areas where fishers from the CGSM claimed to fish based on semi-structured interviews (Torres-Guevara et al. 2016), and with a mangrove-dominated lagoon system in the Sea of Cortez (Carrasquilla-Henao et al. 2013), suggesting an important role for mangroves in artisanal fisheries.

While reported shrimp and crabs catches resembled the spatial distribution of the fishing gear used to catch them, the patterns observed for fish are more interesting as active fishing methods are used in many cases. The spatial distribution of three of the most economically important species (*M. incilis, E. plumieri* and *C. mapale*) were previously modeled by using kriging interpolation in the CGSM (Rueda and Defeo 2001). However, observing the spatial distribution of these species with LEK is a novel approach. Two species reported as important by fishers in our study (*M. incilis* and *E. plimieri*) were also modeled in the study by Rueda and Defeo (2001) and some similar patterns were observed. For example, *M. incilis* was shown to have a higher density in the south of the CGSM during the minor dry season in a similar location where fishers claimed to catch this species. Conversely, *M. incilis* presented a higher density northward in the major rainy season just south of the mouth of the system. The high density areas mapped by Rueda and Defeo (2001) have some similarities to the spatial distribution observed by our LEK approach, whereby *M. incilis* was reported by fishers in the south and at the mouth of the CGSM. Moreover, *M. incilis* density in the rainy season (north of CGSM) was higher which concurs with our results, as that was the area with the highest polygon overlap. Similar patterns were observed for *E. plumieri*, but the distribution obtained from the interviews more closely resembled the density of this species in the dry season. The similarities in the distributions obtained with these different approaches suggest that the distributions of the other species are likely also accurate. While there is a time difference between these studies, the distribution of fishing areas over time has shown to be consistent in the CGSM (this study) and other locations (Turner et al. 2015).

Respondents from Tasajera considered that the catches were similar between 2015 and 2010, but worse in 2005 relative to 2015, whereas most of the remaining respondents considered that catches had declined in both time periods. Catch records for the whole system suggest that there has been a general decline in fish catches from 2005 to the present. However, crustacean catches were similar in 2005 and 2010, but have only shown a negative trend since 2012 (INVEMAR 2017). Fishers from Tasajera and Isla Rosario most frequently caught crustaceans (shrimps and crabs) that have decreased at a slower rate than fish (INVEMAR 2017). However, respondents from Isla Rosario considered that catches have declined in both time periods relative to 2015, despite many respondents claiming that their preferred catch was crustaceans. Fishers from Isla Rosario, however, caught a greater diversity of species compared to those from Tasajera, and therefore the perception of catch declines could be a reflection of overall catches rather than specific groups. The differences across villages may also arise from spatial differences in effort. Records (e.g., in INVEMAR 2017) summarize catches from the whole system although there are villages surrounding the whole system. As a consequence, landings may vary across villages and therefore fishers' perception may differ as well. While fishers' catch perception is not an exact representation of the past, it has concurred with statistical scientific data in other locations (Martins et al. 2018). Unlike catches, the perception of smaller individual sizes in 2005 relative to 2015 was consistent as respondents from all three villages agreed that size of fish had declined over time. Studies in Canada, BC, and Brazil agree with these findings whereby fishers' perception of size shows a negative trend over time mostly caused by high fishing pressure (Eckert et al. 2017; Martins et al. 2018). Records show that high fishing pressure and high juvenile catches (INVEMAR 2017) coupled with a deteriorated ecosystem (Vilardy et al. 2011) have lead to a decrease in catch and size of the resources in the CGSM. Although fishers acknowledged the resource depletion based on our findings, a LEK study in the same area found that they considered that their fishing behaviour was not detrimental to the environment (Torres-Guevara et al. 2016).

4.5.2. Mangroves

Similar to many other regions of the world (Kovacs 1999; Hernández-Cornejo et al. 2005; Dahdouh-Guebas et al. 2006; Ronnback et al. 2007; Hussain and Badola 2010; Moreira dos Santos and Lana 2017) including the Colombian Pacific (Palacios and Cantera 2017), firewood is the most important use provided by mangroves in the CGSM. Mangroves are dense and hard which makes them burn over long periods of time. As a result, they are a valuable and highly used resource across tropical countries (Walters et al. 2008) and, unlike electricity or gas, mangroves are harvested and, therefore cheaper (Palacios and Cantera 2017). Unlike Isla Rosario and Tasajera, fishers from Nueva Venecia used the wood mainly for house construction and poles. These results are not surprising as the Nueva Venecia village is a network of stilt houses located on the water. As such, it was the only village where fishers cut mangroves to use as poles in order to move the canoes around the community and for their fishing activities. Nonetheless, in Isla Rosario and Tasajera the use of mangrove wood for construction is common. Characteristics that make mangroves highly desirable for house construction are the strength of the wood and resistance to rot as they can last for a decade or two (Walters et al. 2008; Moreira dos Santos and Lana 2017). In addition to these advantages, the wood is a freely available resource unlike other construction materials that are likely not affordable for these communities. Consistent with other areas of the world (Kovacs 1999; Dahdouh-Guebas et al. 2006), fishing-related products such as fishing gear (e.g. poles for gillnets) and canoes derived from mangroves are also important across the three fishing villages. While mangrove harvesting is not the fishers' main activity in the CGSM, it is clear that they are consuming mangrove for their livelihood. Unlike other studies (e.g. Kovacs 1999; Dahdouh-Guebas et al. 2006; Palacios and Cantera 2017) we only interviewed male fishers. Our results could change if we included data from other groups in the population. For example, medicinal uses of mangroves have been reported in Africa (Dahdouh-Guebas et al. 2006) and the Colombian Pacific (Palacios and Cantera 2017) but were not reported by any of the respondents in our study. However, the target population of our study was male fishers rather than households. If people other than fishers are using mangroves for other purposes this ecosystem could be under even higher harvest pressure than what was found in this study. While the major threats to mangroves are caused by land cover change and aquaculture (Alongi 2002; Walters 2003), selective harvesting also has detrimental ecological consequences (Walters 2005; López-Hoffman et al. 2006; Palacios and Cantera 2017).

Previous LEK study respondents have mentioned that mangroves are important for fishing productivity (e.g. Dahdouh-Guebas et al. 2006; Moreira dos Santos and Lana 2017; Palacios and Cantera 2017) but our work is the first to include specific questions inquiring about the importance of mangroves for their fishing activities. The fundamental role of mangroves to support fisheries was not in doubt for the fishers interviewed in the CGSM. Though the perception of services that mangroves provide for fish and crustaceans differed by village, all the categories mentioned by fishers have been the subject of scientific research. For example, the role that mangroves play as nursery grounds for reef species on Caribbean islands is widely accepted and understood (Mumby et al. 2004; Nagelkerken et al. 2017). Similarly, field experiments have demonstrated that mangroves support high densities of juvenile fish as the structural complexity of the roots decrease predation risk and create good refuge areas (Laegdsgaard and Johnson 2001). Fishers always identified negative impacts for fisheries in the hypothetical scenario of total absence of mangrove forests. These responses concur with local studies that have explored the mangrove-fishery relationships in similar lagoon systems (e.g. Carrasquilla-Henao et al. 2013) and at a global scale whereby mangrove-fishery relationships have shown a positive trend (Carrasquilla-Henao and Juanes 2017). The similarities between the scientific findings and fishers' perception in the CGSM are remarkable and suggest that fishers knowledge should be included when conducting conservation projects.

Our findings have important implications from a conservation standpoint. Fishers' knowledge of species distribution and in most cases, with respect to catch decline, was consistent with systematic monitoring programs (Rueda 2001; INVEMAR 2017). However, fishery management in coastal lagoon systems such as the CGSM is extremely challenging because their fisheries are commonly multigear and multispecies (Johannes 1998; Rueda et al. 2011), and the jurisdiction is unclear as estuaries are located in the transition zone between sea and land (Blaber 2013; Saavedra-Díaz et al. 2015). Furthermore, there is a discrepancy between the fishing stakeholders and how the fishery management should be conducted. Although fishers are the major resource users they are rarely included in fishery management planning, which has often contributed to ineffective management (Ban et al. 2013). In Colombia, for example, managers' and fishers' perception of adequate management differs in scale because managers have a national perspective while fishers identify issues in their fishing grounds. As a consequence artisanal fisheries are poorly managed (Saavedra-Díaz et al. 2015). These management issues are exacerbated in the CGSM because managers and fishers have different opinions about fishing in the system (Torres-Guevara et al. 2016). In addition to the fishery issues, mangrove-use by locals has not been accounted for in monitoring programs despite the ecological effects that it can induce in the ecosystem. We suggest that if mangrove-use by locals is taken into consideration, the structural analysis of the mangrove will become more robust and conservation plans can be enhanced.

At present, the CGSM is under extreme anthropogenic pressure and highly deteriorated. Fishery resources are decreasing and mangrove mortality has increased while mangrove density has decreased (INVEMAR 2017). Given the importance of the lagoon system, the Colombian government has announced important funding to restore the system. Thus, including fishers in the conservation plans that are underway must be a priority for restoration to be successful. Fishers in the CGSM depend on fishery resources that, in turn, are related to mangroves and other services provided by these trees for subsistence. The detrimental state of the system is directly affecting this vulnerable human population and may perpetuate extreme poverty levels if conservation and restoration plans do not start immediately.

Table 4.1. Frequency (percentage) of responses on possible outcomes in the hypothetical absence of all mangrove coverage in CGSM for each village. IR = Isla Rosario, TA = Tasajera and NV = Nueva Venecia.

	IR	**TA**	**NV**
Catch decline	7.69	4.17	0
Die	5.13	16.67	21.05
Die-water warm up	23.08	0	0
Disappear	2.56	75	31.58
Move to ocean	15.38	0	0
Move to river or ocean	10.26	0	0
Move to rivers	5.13	0	0
Move to sea	2.56	0	0
Move to similar habitat	23.08	4.17	47.37
Stop reproducing	5.13	0	0

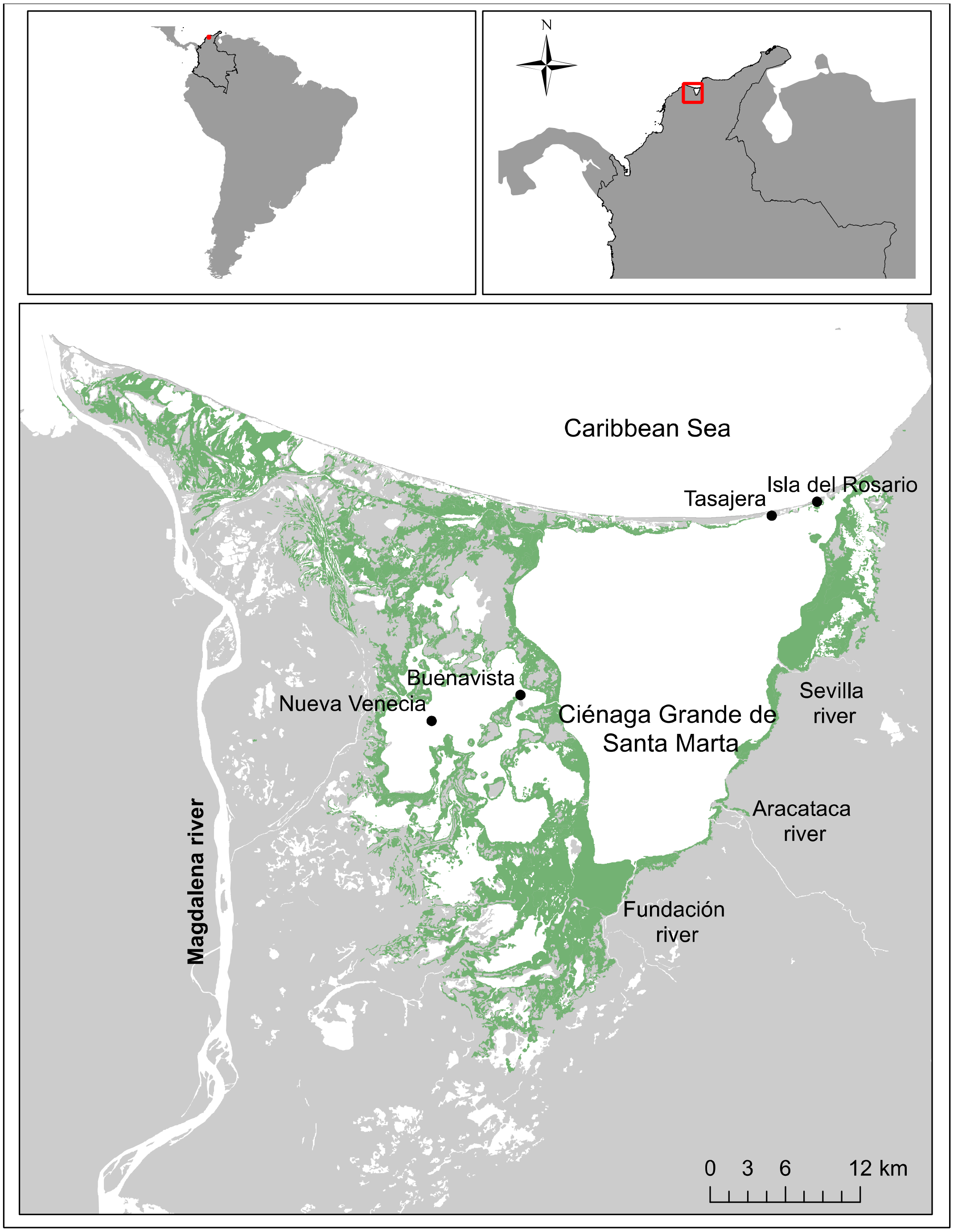

Figure 4.1. Map of the Ciénaga Grande de Santa Marta showing the three fishing villages where interviews were conducted.

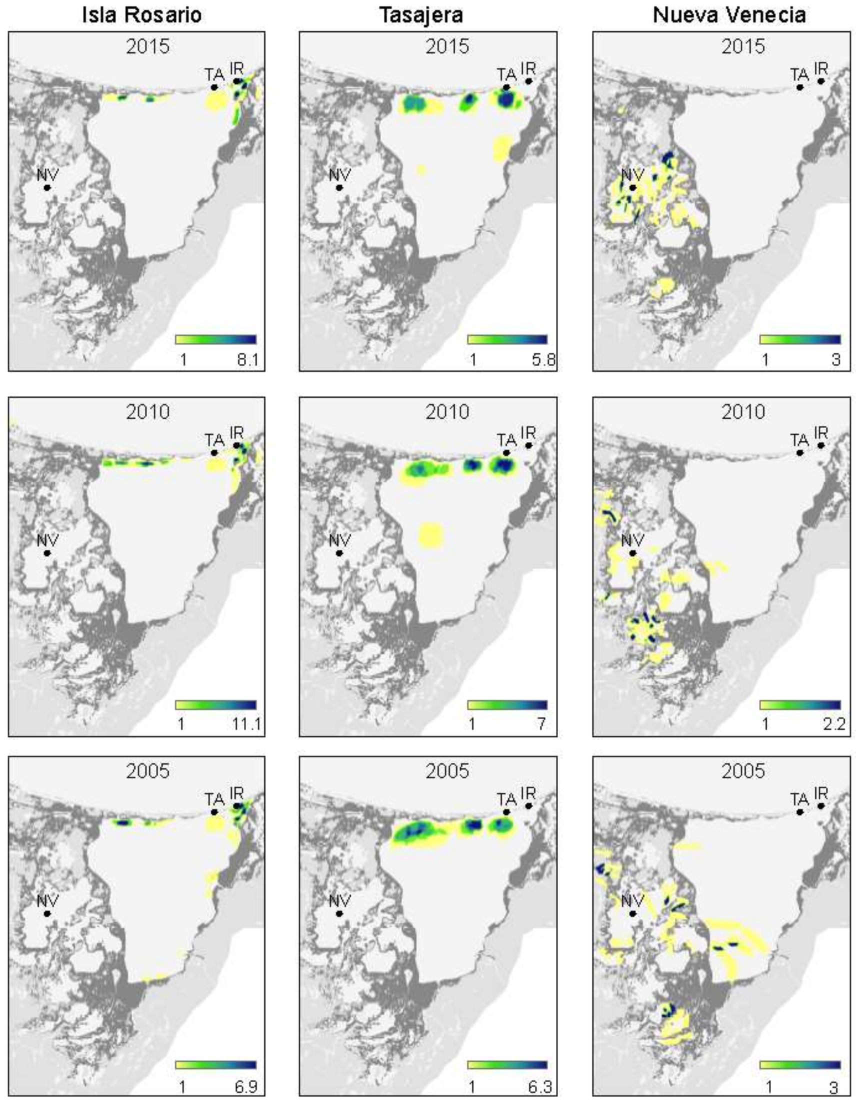

Figure 4.2. Maps showing fishing areas across villages and over three time periods, recent (2015), 5 years ago (2010) and 10 years ago (2005). In the legend, yellow colours represent no polygon overlaps whereas blue represents high polygon overlap. Each map has its own scale as the number of overlapping polygons differs across all combinations. In total 82 fishers were interviewed (n = 39 in Isla Rosario (IR), n = 24 in Tasajera (TA) and n = 19 in Nueva Venecia (NV).

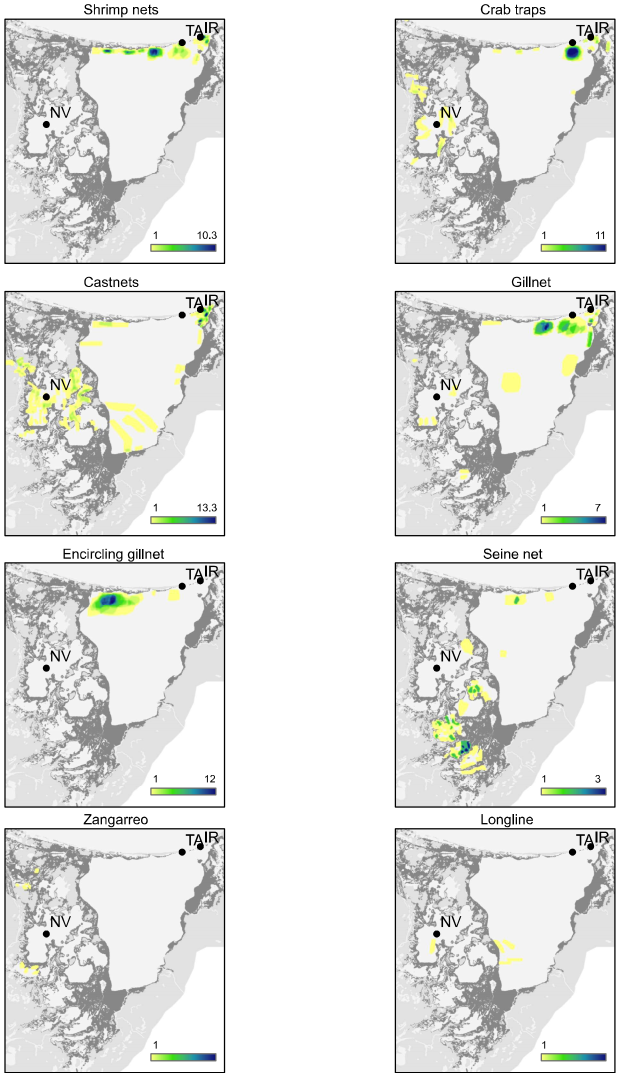

Figure 4.3. Maps showing the spatial distribution of the different fishing gears combined for all villages used by fishers interviewed in IR (Isla Rosario, n = 39), TA (Tasajera, n = 24) and NV (Nueva Venecia n = 19). In the legend, yellow represents low (no polygon overlap) while blue represents high (many polygons overlapping) occurrence. Each map has a unique scale because the polygon count was done separately for each gear and thus is related to number of respondents. There are no upper limits for zangarreo and longline use as there were no overlapping polygons.

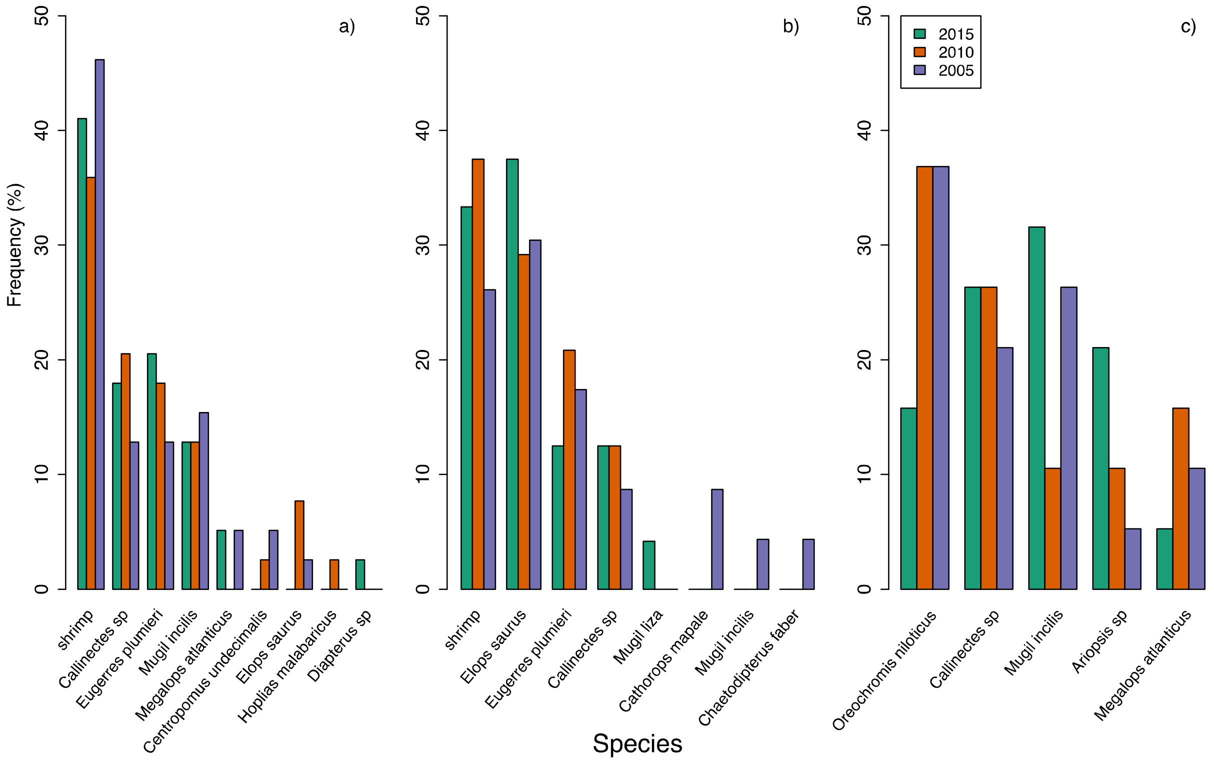

Figure 4.4. Frequency (percentage) of the most abundant species caught at three different time periods (2015, 2010 and 2005) in a) Isla Rosario (IR) b) Tasajera (TA) and c) Nueva Venecia (NV) according to the fisher's responses in each village.

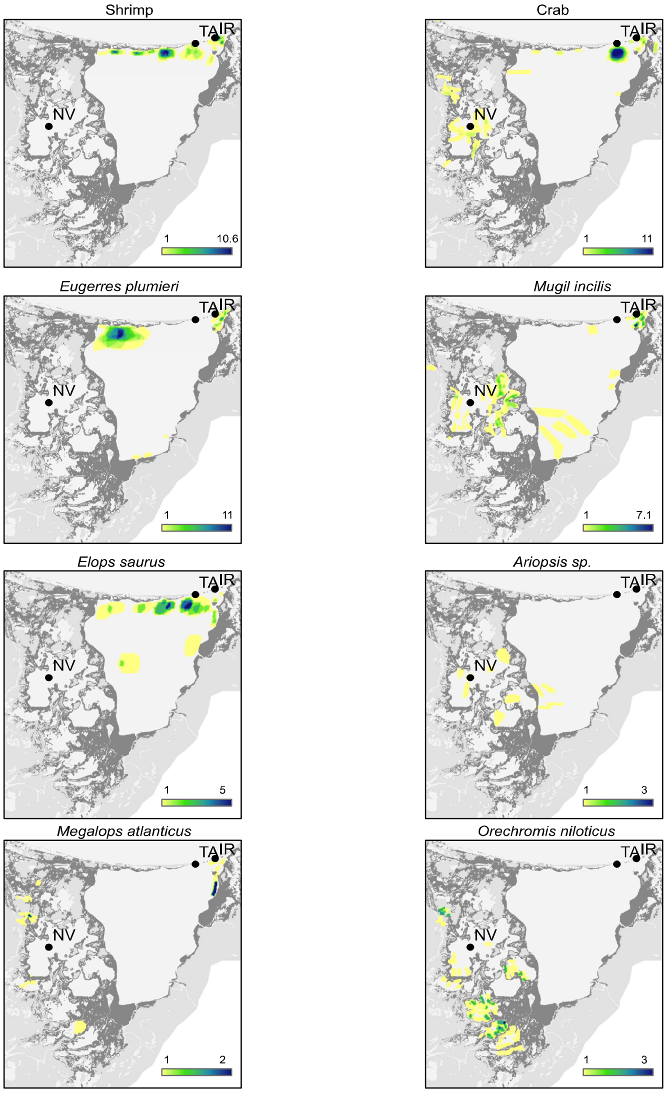

Figure 4.5. Map showing the spatial distribution of catches of the most frequent species caught by fishers interviewed in IR (Isla Rosario, n = 39), TA (Tasajera, n = 24) and NV (Nueva Venecia, n = 19). In the legend, yellow represents low (no polygon overlap) while blue represents high (many polygons overlapping) occurrence. Each map has a unique scale because the polygon count was done separately for each species and thus is related to number of respondents.

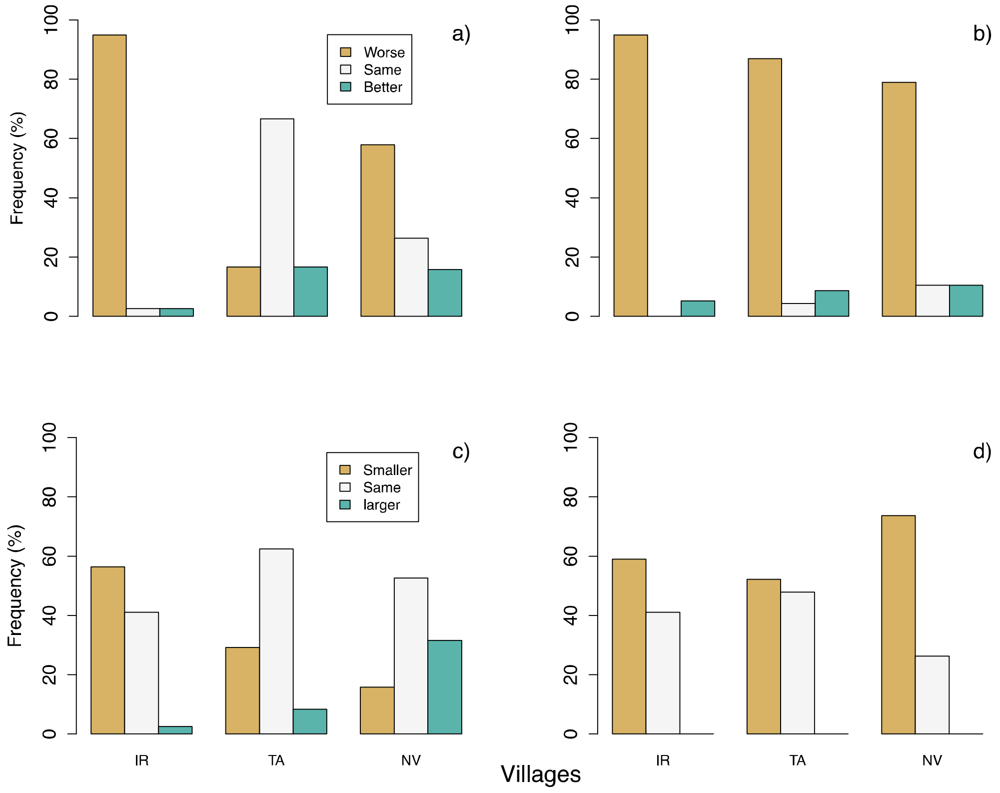

Figure 4.6. Frequency of fishers' perception in 2015 relative to 2010 (a and c) and 2005 (b and d) for catch (a and b) and organism size (c and d) of the most abundant species caught in the three time periods for the three fishing villages where interviews were conducted. IR = Isla Rosario, TA = Tasajera and NV = Nueva Venecia.

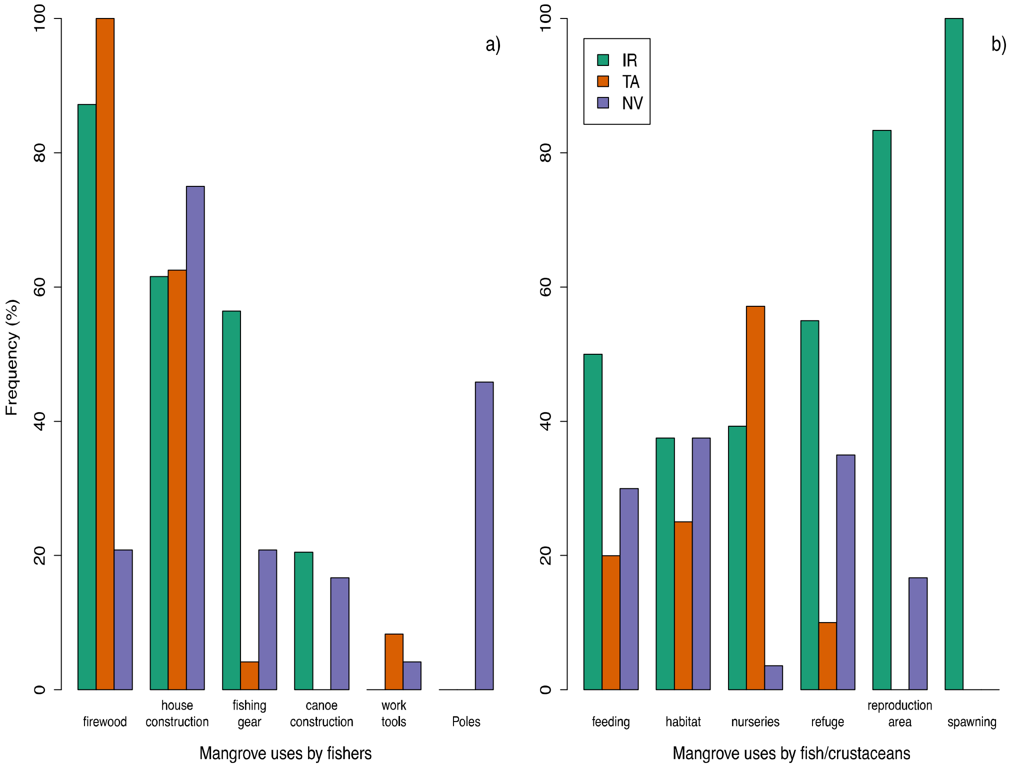

Figure 4.7. Frequency (percentage %) of mangrove uses by a) fishers and b) crustaceans and fish according to respondents from three fishing villages. IR = Isla Rosario, TA = Tasajera and NV = Nueva Venecia. In a) Most respondents mentioned more than one use and thus the percentages are calculated based on the number of individuals in each fishing village who mentioned each given category.

Chapter 5 - Predicting important rockfish (*Sebastes* spp.) habitat from large-scale longline surveys for southern British Columbia, Canada

Adapted from: Mauricio Carrasquilla-Henao[1], K. Lynne Yamanaka[2], Dana Haggarty[2] and Francis Juanes[1](2017). *Canadian Journal of Fisheries and Aquatic Sciences*. Accepted pending revisions.

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5.2. Abstract

Rockfish, particularly Yelloweye, and Quillback, are vulnerable to overfishing because they mature late and have affinity for shallow water (50-200m) habitats. Because studies relating habitat characteristics with the distribution and presence of rockfish at large scales remain scarce we sought to investigate the relationships between benthic characteristics with the presence-absence and abundance of rockfish caught in longline surveys in nearshore waters of southern British Columbia. Habitat parameters were calculated from a 20m resolution bathymetry layer. Yelloweye, and Quillback rockfish were examined separately, and combined with 19 other rockfish species in a species aggregate (total rockfish); occurrence data were fitted with generalized linear mixed effects models and abundance data were fitted with zero-inflated mixed effects models. Presence-absence and abundance were positively associated with increasing slope and latitude, whereas distance to rock and fine bathymetric position index (FBPI) were negatively associated. These relationships demonstrate the importance of rock and should be accounted for when considering spatial rockfish conservation measures. While Underwater visual observations (UVO) data offer visual fish habitat and abundance, longline surveys may be a more cost-effective method for large-scale studies.

5.3. Introduction

Anthropogenic activities can cause water pollution, habitat degradation, and overfishing of many fish populations (Worm et al. 2006; Halpern et al. 2008; Worm et al. 2009) in many marine ecosystems. Consequently, a decline in biodiversity and ecosystem services has been reported in degraded marine systems (Worm et al. 2006). To date, the most conservative conservation strategy for protecting both the ecosystem and fish populations are marine protected areas (MPAs), whereby detrimental anthropogenic activities, such as fishing, are strictly controlled or forbidden (Halpern 2003; Edgar et al. 2014). MPAs can be an effective spatial management tool that serves to increase population size, individual fish size, species diversity, and protect critical habitats and enhance adjacent fisheries (Roberts et al. 2001; Haggarty et al. 2016b).

The relationship between organisms and their environment has been a key focus in ecology (Rushton et al. 2004). The development of species distribution models (SDM) to tackle this relationship has increased our knowledge of how species are distributed in the landscape, over space and time (Rushton et al. 2004; Elith and Leathwick 2009), and what important environmental factors best suit their biological needs. Moreover, SDM have been extremely helpful for ecosystem management and conservation as spatial planning tools to create protected areas in the marine environment based on the predictive outcomes of the models (Le Pape et al. 2014).

Until recently, marine benthic habitats were described by direct observations of the environment and thus, were limited to shallow waters over site specific or limited spatial scales due to logistical constraints. In the last few decades new remote sensing technologies have allowed researchers to map the marine floor over large spatial extents and at very high spatial resolution (Kostylev et al. 2001; Young et al. 2010) facilitating the prediction of species distributions and abundance in the marine environment (Rubec et al. 1998; Pittman and Brown 2011; Young and Carr 2015; Rubec et al. 2016). As part of a strategy to recover the overfished populations of Yelloweye rockfish (*Sebastes ruberrimus*) and Quillback rockfish (*S. malliger*) in nearshore waters of British Columbia (BC), Canada, 164 Rockfish Conservation Areas (RCAs) were established in 2007 that are closed to industrial fishing although some fishing is allowed with specific gears. The boundaries were created based on the best-available bathymetry data at the time (100m) together with georeferenced catch data from recreational and industrial fishing (Yamanaka and Logan 2010). In BC and Puget Sound in Washington State waters, rockfish abundance has declined considerably due to overfishing by hook and line, longline, and trawl fisheries (Williams et al. 2010; Yamanaka and Logan 2010).

Several studies have quantified the abundance of rockfish in relation to habitat using visual, *in situ*, methods employing Remotely Operated Vehicles (ROV) (Du Preez and Tunnicliffe 2011; Wedding and Yoklavich 2015; Haggarty et al. 2016b) and SCUBA diving (Marliave et al. 2009). More recently, these visual and SCUBA-derived abundance estimates have been paired with bathymetry layers from high-resolution (3 m) bathymetry data on California's central coast (Young et al. 2010; Young and Carr 2015). While these studies have been informative about how rockfish use their habitat, most of them are limited to small geographic locations generally due to operational costs. However, large spatial extent studies are rare for groundfish species in general, and for rockfish in particular. An alternative approach to visual survey data is to use fishery independent research longline catch data collected over multiple years and large spatial extents (100s of km) to address these species distribution questions.

In this study, we combine medium resolution (20m) remote sensing imagery, and longline survey catch data within a GIS framework to quantify relationships in fish presence-absence and abundance with habitat parameters using Generalized Linear Mixed Models (GLMMs) for presence-absence data and zero-inflated models for count data. To our knowledge this is the first study to analyze habitat characteristics for rockfish and large spatial extent scales using fishery independent data and digital elevation models. The goal of this study is twofold; i) analyze possible relationships between benthic habitat characteristics derived from a GIS framework with presence and abundance of two species of rockfishes, Yelloweye, and Quillback rockfish, caught in longline surveys and ii) evaluate the predictive potential of species distribution models at large spatial extents derived from fishery-independent data. We hypothesis that rock reef habitat identified by complexity features will predict higher presence and abundance of rockfish.

5.4. Methods

5.4.1. Fish sampling:

We used annual research longline survey catch data collected by Fisheries and Oceans Canada (DFO) as part of the Inshore Rockfish research and assessment program. Longline surveys are conducted in inside waters of southern BC for Yelloweye, and Quillback stock assessments (Fig. 5.1). For the purpose of this study we used all the available survey data that were collected from 2003 to 2015, excluding 2006, when a survey was not conducted. Fish were sampled using a depth-stratified sampling design that standardized surveys between shallow (41-70m) and deep (71-100m) depth intervals. Each year, a pre-determined number of 2km x 2km survey blocks for both shallow and deep depth intervals are randomly selected and one longline set was fished within each selected block. When deploying the longline fishing gear, the aim was to fish on hard bottom to target rockfish (Yamanaka and Lacko 2004).

Each longline (~550m in length) consists of 225 circle hooks baited with ~30g of squid, each separated by approximately 2.67m (8 feet) a long a weighted groundline that is anchored at each end. The longline gear used is similar to that used by the commercial fishery for rockfish and hence is selective for larger, adult fish. The longline is soaked for two hours starting from the time of the last anchor deployed over the side of the fishing vessel when setting the gear, until the first anchor was hauled on board the fishing vessel to retrieve the gear. Start and end points are recorded by the vessel's GPS system when the first and last anchor are set over board (Yamanaka and Lacko 2004). All rockfish were weighed (g) and measured (total length, cm), dissected to determine sex and gonad maturity determined, and tissue and otolith samples taken for DNA and age analysis, respectively. Thus, for each set, presence or absence, and abundance of each species can be determined.

We created a presence-absence and abundance datasets for all rockfish species caught, and separately for Yelloweye, and Quillback (Table 5.1). We also examined Yelloweye, and Quillback independently because they are the most valuable and most abundant species in the commercial fishery. As a consequence, there has been considerable population declines due to overfishing in BC (Yamanaka and Logan 2010).

5.4.2. Spatial data collection and processing

A 20m resolution Digital Elevation Model (DEM) was used to obtain six different raster layers (see Haggarty 2015 for complete methods), curvature, slope, rugosity and bathymetric position index (see definitions in Table 5.2) at 3 different spatial scales, fine, medium and broad. To include latitudinal effects due to the extent of our study area, we created a latitude raster layer by Kriging interpolation using the start latitudes from the sets surveyed. This latitude layer was created with the same spatial resolution as the rest of the raster datasets (i.e. 20m). Finally, we calculated Euclidean distances to rock and mixed substrate using a substrate model (Haggarty 2015) and produced two new raster layers (Table 5.2).

In order to sample the raster layers at each longline location, we first connected start and end coordinate points to simulate the line by using the "*XY to line*" tool in Arcmap 10.1. Next, we used "*zonal statistics*" to obtain the mean value of each explanatory variable under the line. We used this approach for two main reasons; first, because the length of a longline is greater than one pixel (20m) and thus, sampling one unique pixel would not reflect the habitat characteristics under the entire line. Second, because it provides a better habitat description for all the fish caught on the longline. We also calculated the area of the four substrates in the substrate model (i.e. rock, mixed substrate, sand and mud) with the "*tabulate area*" tool. A new dataset with the mean value per site of each explanatory variable was created, together with the area of each substrate.

When overlaying the longline datasets with the bathymetry layer, some sites appeared to be on land rather than in the water (i.e. bathymetry values > 0). To avoid this problem we eliminated all sets with average depth values < 20 m and thus our analysis included 743 sets across all years.

5.4.3. Statistical analyses

5.4.3.1. Presence-absence:

A site was classified as a 'presence' if there was at least one rockfish, or Yelloweye or Quillback while an 'absence' occurred when there were no rockfish caught on a longline. The presence-absence data were analyzed in two separate steps. First, we used GLMMs to obtain the best model and secondly, we used a Generalized Linear Model (GLM) with the same terms of the GLMM but excluding the random effect (year) to construct a predictive raster. GLMs do not require linearity or constant variance, and allow response variables that have a number of different distributions (e.g., Poisson, negative binomial, binomial). Because of this flexibility GLMs have proven to be a powerful statistical tool for modeling species distributions (Guisan et al. 2002).

Our dataset was randomly divided into two separate datasets, 70% (n = 520) as training data and 30% (n = 223) as test data. Training data were used to model the response variable with the predictors while test data were used to evaluate the accuracy of the models (Young et al. 2010). We modeled total rockfish, Yelloweye, and Quillback separately using GLMMs with a binomial distribution and logit link function (Zuur et al. 2009) and included year as a random effect to account for the structure in the data (i.e.

many sites sampled within a year). In order to obtain standardized effect sizes, explanatory variables were scaled and centered to have a mean of 0 and a standard deviation of 1 (Legendre and Legendre 2012). Prior to the modeling process, collinearity among predictor variables was tested. A global model including all explanatory variables was created and alternative nested models derived from the global model were used to determine the best model based on Akaike's Information Criterion (AIC). The model with the lowest AIC value was considered the best model. If two or more models were within 2 units of the best model, the most parsimonious (i.e. fewest explanatory covariates) was chosen (Burnham and Anderson 2003). Once the best model was selected the Variance Inflation Factor (VIF) was computed to corroborate that the predictors within a model were not collinear (Zuur et al. 2009). The variance explained (R^2) was calculated based on Nakagawa and Schielzeth (2013) method which mathematically differs from R^2 in linear models, but provides an accurate estimation of the variance explained in GLMMs. All GLMMs were fitted using the *lme4* package (Bates et al. 2015) in R (R Core Team 2014).

The Marine Geospatial Ecology Tools (MGET) in ArcGIS is a toolset that incorporates ArcGIS with the R statistical package to create predictive raster layers from the modeling process (Roberts et al. 2010). However, one limitation of the toolset is that it does not run GLMMs. To overcome this problem, when the best GLMM model was obtained, we removed the random effect and calculated a GLM with the same terms retained by the mixed model. In all cases there were no differences in the significance or direction of the relationship in the GLMM vs. the GLM (Fig. A5.1). We ran the GLM model with the training data and then created a predictive raster. For each species, the raster dataset was reclassified with values of 1 (presence) and 0 (absence) with a cutoff of 0.7. These cutoffs are within the ranges used for similar species (Iampietro et al. 2008; Young et al. 2010). These cutoffs mean that values above 0.7 for total rockfish, Yelloweye and Quillback were classified as presence (1) while values below the cutoff were classified as absence (0). To test the accuracy of the model we calculated the agreement between the training data and the test data with Cohen's Kappa statistic within the MGET toolset in ArcGIS (Roberts et al. 2010; Young et al. 2010). Kappa index ranges from 0 to 1 whereby 1 is a perfect agreement between train and test data while 0 represents no agreement between both data. Values in between represent slight agreement (0 – 0.2), fair (0.21– 0.40), moderate (0.41 – 0.60), substantial (0.6 – 0.80), and almost perfect (Landis and Koch 1977).

As part of the model validation process we used semi-variograms to detect spatial autocorrelation in the residuals of the best model for each species (Fig. A5.2 and Fig. A5.3). Spatial autocorrelation occurs when close values are more similar than values further apart. If spatial autocorrelation occurs there is a violation of the assumed independence in the data and the spatial structure must be accounted for to prevent this issue (Legendre 1993).

5.4.3.2. Abundance:

The number of total rockfish, Yelloweye, and Quillback were counted for each line. We used zero-inflated negative binomial (ZINB) models (Martin et al. 2005) to predict abundance from the explanatory variables because there was an excess of zeros in the three datasets (24%, 52.5% and 32.8% of sets for total rockfish, Yelloweye, and

Quillback rockfishes, respectively). Zero-inflated models have been demonstrated to improve the fit relative to Poisson or negative binomial distributions in the presence of excess of zeros (Potts and Elith 2006). We selected zero-inflated over hurdle models because we wanted to distinguish between false zeros and true zeros. False zeros occur when the habitat is suitable but a fish is not found while true zeros can occur when the habitat is not suitable (Martin et al. 2005; Potts and Elith 2006).

As above, in presence-absence models, collinearity between predictors was examined prior to running the models and to ensure that highly correlated explanatory variables were not included in the same model. Prior to analysis the predictive variables were standardized to have a mean of zero and a standard deviation of one. A global model including all the predictors and including year as a random effect to account for the many sites sampled within one year was constructed, and a number of nested models were built. The best model for each dataset was determined by selecting the lowest AIC. Finally, the model validation process included a spatial autocorrelation analysis using Moran I correlograms as well as a visual inspection of the residuals against fitted values and each explanatory variable (Zuur et al. 2009; Zuur et al. 2010). All models were constructed in the *glmmAMD* package in R (Skaug et al. 2016).

5.5. Results

5.5.1. Presence-absence

For total rockfish nine predictors were retained in the best model, including slope, distance to rock and latitude (Fig. 5.2a). The probability of occurrence of any species of rockfish was positively related to curvature (1.04 95% CI: 1.61, 0.49), slope (0.92 95% CI: 1.37, 0.52) and latitude (0.66 95% CI: 0.94, 0.41) while negatively correlated with mud area (-0.32 95% CI: -0.08, -0.56), distance to rock (-0.37 95% CI: -0.14, -0.61) and FBPI (-0.37 95% CI: -0.47, -1.3) (Fig. 5.2a). We obtained a Kappa value of 0.46 for the agreement between the test and train data. While the model performed better predicting presences correctly (80%) rather than absences (69.7%) and was also more accurate in the center of the study area (Johnstone Strait and nearby inlets) where a large proportion of presences were accurately predicted (Fig. 5.3a).

Unlike the total rockfish best model, the best Yelloweye rockfish model did not retain curvature and mud covariates. However, some of the terms retained for Yelloweye have similar relationship patterns to total rockfish. Slope and latitude were positively correlated with the probability of occurrence, but distance to mix substrate also had a positive, but small, effect size (0.27 95% CI: 0.0039, 0.53), unlike in total rockfish. In contrast, distance to rock substrate, FBPI and rugosity were negatively correlated. Rugosity was significant for Yelloweye rockfish while non-significant for total rockfish and the distance to rock effect size was larger in Yelloweye rockfish (-0.85, 95% CI: -0.43, -1.38) compared to total rockfish (-0.37, 95% CI: -0.15, -0.61) (Fig. 5.2b). The Kappa value for the agreement between the train and test data was 0.45. Unlike for total rockfish, Yelloweye rockfish had a lower percentage of presences correctly predicted (68%) compared to absences (76%). The model also had the majority of presence agreements in the inlets in the center of the study area (Fig. 5.3b).

The best model obtained by AIC for Quillback rockfish showed that curvature, slope and latitude were positively correlated, while FBPI, sand, mud and distance to rock had a negative relationship (Fig. 5.2c). The latitude effect was larger (0.77, 95% CI: 1.02, 0.53) than for total rockfish (0.66, 95% CI: 0.94, 0.41) and Yelloweye (0.55, 95% CI: 0.94, 0.21) while the distance to rock was significant, but the effect size was small. As for total rockfish and Yelloweye rockfish, depth was not significant. The agreement between the train and test data based on the Kappa index was 0.49. Presence was correctly predicted 71.6% of the time whereas absence was predicted correctly 81.7% of the time. The greatest agreement between longline data and the model output was in Johnstone Strait resembling the patterns observed for Yelloweye rockfish and total rockfish (Fig. 5.3b). In all three cases, and according to the predictors, there is more suitable habitat in northern waters and inlets of the inshore waters separating Vancouver Island from the mainland (Fig. 5.3).

5.5.2. Abundance:

Unlike for the presence-absence models, positive but weak relationships were observed between depth and total rockfish, Yelloweye, and Quillback. Latitude and slope showed a positive relationship for total rockfish, but latitude had a stronger effect (0.42 vs 0.16). In contrast, distance to rock had a strong negative effect (-0.44 95% CI: -0.58, -0.29) for total rockfish while FBPI had a weak effect (Fig. 5.4a).

Yelloweye abundance (mean = 2.52 ± 4.43(sd)) increased with slope, as distance to mixed substrate increased and northward, while abundance decreased as distance to rock increased. Two main covariates – latitude and distance to rock - predicted Quillback abundance, whereby abundance increased with latitude and decreased with increasing distance to rock (Fig. 5.4c).

Quillback had twice the mean number of individuals per set (5.37 ± 7.99(sd)) than Yelloweye. Although abundance increased with slope, the effect of slope for Quillback (0.09 95% CI: 0.09 – 0.12) was smaller than for Yelloweye (0.39 95% CI: 0.26 – 0.51). However, latitude effects were larger for Quillback compared to Yelloweye (Fig. 5.4).

Latitude was one of the most important abundance predictors of rockfish and, in all datasets, abundance increased with latitude. However, Yelloweye abundance peaked in the inlets of Johnstone Strait while Quillback abundance continued to increase northward (Fig. 5.5). The maximum number of Yelloweye (31), and Quillback (56) per set also differed. However, the geographical location where the peak abundance was found was similar, but large numbers of Quillbacks per set were also found further north (Fig. 5.5b and 5.5c).

5.6. Discussion

We found strong evidence of the importance of habitat complexity for both presence and abundance data for total rockfish, Yelloweye, and Quillback. While there were some differences in the predictors obtained within presence and abundance models, there are similar patterns across both approaches. For example, slope (except for Quillback in zero-inflated models), FBPI, distance to rock and latitude were consistently important predictors within species and model types. The most noticeable difference between the two analyses was the lack of significance of depth in the presence-absence approach, but its positive correlation in the abundance dataset. A possible explanation for the lack of significance in the former models could be related to the sampling designed used in the longline surveys. The sets were divided into shallow (41 -70m) and deep (71 -100m) which does not span the entire depth range of Yelloweye (up to 500m) or Quillback (up to 275m) (Love et al. 2002) nor does it include the variable depth ranges of the other rockfish species included in the total rockfish category. In contrast to these findings, Young et al. (2010) found depth to be a significant predictor for the probability of occurrence of three different rockfish species in Cordell Bank, California using the same statistical approach as we did. However, a broader depth range was sampled in that study compared to ours. When predicting abundance, depth had a positive effect, though the magnitude was small, in all three cases. These findings are consistent with the results obtained by Young and Carr (2015) using visual surveys in California for various rockfish species, and by Williams and Ralston (2002) using trawl surveys in California and Oregon for a number of groundfish including rockfish. Similar to Young and Carr (2015), predictors other than depth were more important in our models. This consistent pattern suggests that, despite the sampling design issue, depth may be a better predictor for fish size, since many rockfish species migrate from shallow nursery habitats to deep habitats as they grow (Love et al. 1991; Johnson et al. 2003; Yamanaka et al. 2006a; Yamanaka et al. 2006b). Unfortunately, due to our gear selectivity we were unable to conduct size analyses to confirm or reject this pattern.

When comparing across the three scales of bathymetric position indices (FBPI, MBPI, and BBPI) included, in all cases the best models only retained FBPI, which was negatively correlated with occurrence and abundance. These results suggest that the habitat characteristics for total rockfish, Yelloweye, and Quillback are better described at fine scales relative to the bathymetry resolution (20m). While bathymetric position indices at different scales describe different habitat classes (Young et al. 2010), our results suggest that habitat complexity is present (Lundblad et al. 2006) and related to rockfish occurrence and abundance within inshore waters of southern BC.

Areas with high slopes better predicted the occurrence of the three species modeled. Similarly, abundance of rockfish tended to increase as slope increased. However, the magnitude of the effect was largest for Yelloweye, while barely significant for Quillback. UVO observations of Yelloweye, and Quillback have found them on vertical walls in complex habitats in Alaska (Johnson et al. 2003), while other rockfish species have also been associated with vertical habitats in Monterrey Bay, California (Yoklavich et al. 1999) and BC (Richards 1986) likely because many of these habitats have crevices that are used as refuges (Yoklavich et al. 1999). Given the geology of our study area, it is likely that the areas of high slope are areas of rock walls, and while not visible with remote sensing techniques, our results were consistent with studies conducted with UVO observations in similar areas in BC and California.

Distance to rock was negatively correlated to presence and fish abundance for all three cases in both presence-absence models and zero-inflated models. These consistent correlations suggest that the probability of occurrence and abundance of rockfish in general and Yelloweye, and Quillback in particular, decrease as the distance to rock increases. Young et al. (2010) found a similar relationship for *S. flavidus* and *S. rosaceus* in Cordell Bank, California. Similarly, many studies conducted by UVO have found that many species of rockfish are associated with boulder and bedrock in larger densities than in less complex habitats (O'Connell and Carlile 1993; Yoklavich et al. 1999; Johnson et al. 2003). Again, despite the two different techniques, UVO and derived bathymetric layer, results are consistent between techniques.

Both the probability of occurrence and abundance increased with latitude for the three datasets. Although the predictive raster layers (Fig 5.3) show suitable habitat in the south, the probability of occurrence was low. Similarly, abundance for total rockfish, Yelloweye, and Quillback were lower in the south where many longlines failed to catch rockfish. However, the pattern differed between Yelloweye, and Quillback rockfish in more northern latitudes where Quillback's abundance tended to continue to increase northward, but Yelloweye abundance peaked in Johnstone Strait inlets (Fig. 5.5), hence the larger effect size of latitude for Quillback with respect to Yelloweye. One possible explanation for this pattern is that latitude is correlated with other important oceanographic variables (Young and Carr 2015) not accounted for in this study. Indeed, the inshore waters of southern BC have complex oceanographic dynamics (Burd et al. 2008) driven by fresh water flows (e.g. Fraser river) and tidal regimes that could explain such patterns. However, studies in Alaska have shown that Yelloweye, and Quillback are tolerant of fluctuating oceanographic conditions (Johnson et al. 2003). An alternative, and perhaps more plausible explanation, is that southern Vancouver Island and the mainland have a higher human population than northern latitudes which may have negatively impacted rockfish populations by exerting a larger fishing pressure and/or other types of human stressors (Frid et al. 2016). Such patterns have also been suggested in Alaska's inshore waters to explain rockfish distribution (Johnson et al. 2003).

Kappa values between 0.41 and 0.60 suggest moderate agreement between training and test data (Landis and Koch 1977). All our Kappa values for the presence-absence models fell within this range, though were lower than those measured in Young et al. (2010) analysis of California rockfish species, that obtained substantially larger values and almost perfect agreement for *S. flavidus* and *S. rosaceus* respectively. However, our results for the dataset evaluation are similar to those of Iampietro et al. (2008) (0.421), also In California. Although our Kappa values are smaller than those in Young et al. (2010), we consider that our approach has several promising advantages moving forward and that the agreement values we obtained are reasonable based on large scale longline data used. The limitation of our approach assumes that the longline was set completely straight along the bottom substrate by connecting the start and end coordinates in our GIS software. However, it is likely that this was not the case and that some variation from linear did occur while setting the longline on the bottom. Because we did not have hook by hook data for each longline, we were unable to relate each hook to a specific pixel in the bathymetry and derived layers. In contrast, Young et al. (2010) sampled with an Remotely Operated Vehicle (ROV) and could obtain exact coordinates for the presence and absence of each fish and relate them to high resolution bathymetry data. To overcome this problem we estimated an average of each layer under the assumed straight line. A final major difference with respect to species distribution models conducted in California is the spatial resolution of the raster layers. While our layers had a 20m resolution, studies in California, using ROVs, have used higher resolution (3m and

5m) to assess rockfish distributions, biomass and density (Young et al. 2010; Wedding and Yoklavich 2015; Young and Carr 2015). An advantage of our study is that we were able to assess rockfish distribution and abundance at a much larger spatial extent relative to studies conducted with ROVs or visual surveys. We have also taken advantage of existing information collected from fishery independent surveys conducted for stock assessment purposes. While large spatial extent studies could be possible with ROVs a strong limitation to this approach are the high costs of transporting and operating the vehicle compared to obtaining the fishery independent data. However, studies conducted with submersibles or by SCUBA diving can classify rockfish habitats as biogenic or abiotic (Du Preez and Tunnicliffe 2011), thus providing additional resolution to habitat predictors. Nonetheless, the development of algorithms that accurately predict some of the habitat characteristics observable with visual techniques has improved considerably and has helped to characterize habitat based on indirect data.

Our results are important for rockfish conservation and marine spatial planning in BC waters. Our results identify suitable habitat where rockfish occur and potential habitat for population recovery purposes, and reflect the spatial distribution and abundance patterns of commercially important species. This information can be used to create new RCAs or reevaluate and reconfigure existing RCAs in terms of their spatial location to account for probability of occurrence and abundance. To date RCAs in inshore waters of southern BC have not contributed substantially to the recovery of rockfish populations in the southern areas of the coast (Haggarty et al. 2016b). However, there is some evidence of recovery in BC's central coast (Frid et al. 2016). Although on the BC southern coast there is lack of compliance by recreational fishers to RCA regulations that may be affecting rockfish population recovery (Lancaster et al. 2015; Haggarty et al. 2016a; Lancaster et al. 2017), it is also possible that RCAs may need more time to show recoveries due to long-lived rockfish life histories. Our results provide critical habitat information that may be used for future spatial marine planning to help recover rockfish populations. These findings also contribute to better describe rockfish habitat, which is a critical aspect of Ecosystem Based Fishery Management (EBFM) in order to better manage target species, their habitat and the species that interact with them (Pikitch et al. 2004). These models, and species distribution models in general, can have a positive impact on EBFM by expanding the knowledge we have on the systems target species use.

In conclusion, our study is one of the first studies to integrate multi-year fishery independent data and remote sensing imagery at large spatial extents (100s km) to predict rockfish probability of occurrence and abundance. We found that total rockfish, Yelloweye, and Quillback are strongly related to habitat heterogeneity in inshore waters of southern BC. These findings confirm the importance of rock and structured habitat for demersal species of the genus *Sebastes*, particularly Yelloweye, and Quillback. The moderate agreement (Kappa values) between test and train data highlights the convenience of using more inexpensive sampling techniques to sample large spatial extents and yet have accurate models to use for spatial planning and conservation purposes. Overall, our results are critical for marine spatial planning and conservation decision makers in Canada, and will help efforts to improve rockfish spatial management and conservation.

Table 5.1 List of rockfish species and total fish caught across all sampling years included in the study.

Rockfish Species	Total abundance
Black	12
Blue	4
Canary	150
China	11
Copper	265
Greenstripped	177
Harlequin	2
Quillback	4286
Redstripe	10
Rosethorn	6
Sharpchin	2
Silvergray	19
Tiger	60
Vermelion	4
Widow	1
Yelloweye	2045
Yellowtail	54

Table 5.2 Description of the explanatory variables derived from the digital elevation model and calculated from the substrate model (Haggarty 2015) that were included in the modeling process for both the presence-absence and abundance analysis.

Variable	Definition	Spatial resolution
Depth (m)	Obtained from a digital elevation model. Raster dataset that contains elevation values for each pixel.	20m
Curvature	Rate of change of the bathymetric slope (Haggarty 2015). Negative values indicate a convex surface at that pixel, a positive value indicates a concave surface at that pixel and a 0 indicates a flat surface.	20m
Slope	Rate of change in depth between any given cell and the neighbor cells. Values are in degrees, the higher the value the higher the slope.	20m
Rugosity	Index of surface complexity calculated by dividing the contoured distance by the planar distance (Du Preez 2014). Values	20m

	range between 0 (flat terrain) and 1(high roughness).	
Bathymetric Position Index (BPI MBPI FBPI)	BPI captures large characteristics within the landscape, while MPI (medium BPI) captures features at a lower scale and FBPI (fine BPI) captures fine scale bathymetric characteristics (Wright et al. 2012; Haggarty 2015). Negative values describe valleys, positive BPI values are ridges, while 0 values are either flat surfaces or constant slopes.	20m
Latitude	A raster dataset that covers the entire study area where each cell has a latitude value.	20m
Distance to rock	A raster dataset that calculates the distance to rock substrate. Values are in meters.	20m
Distance to mix	A raster dataset that calculates the distance to mixed substrate. Values are in meters.	20m
Substrate area (rock, mixed, sand and mud)	Area in m^2 of rock, mixed substrate, sand and mud under each line.	NA

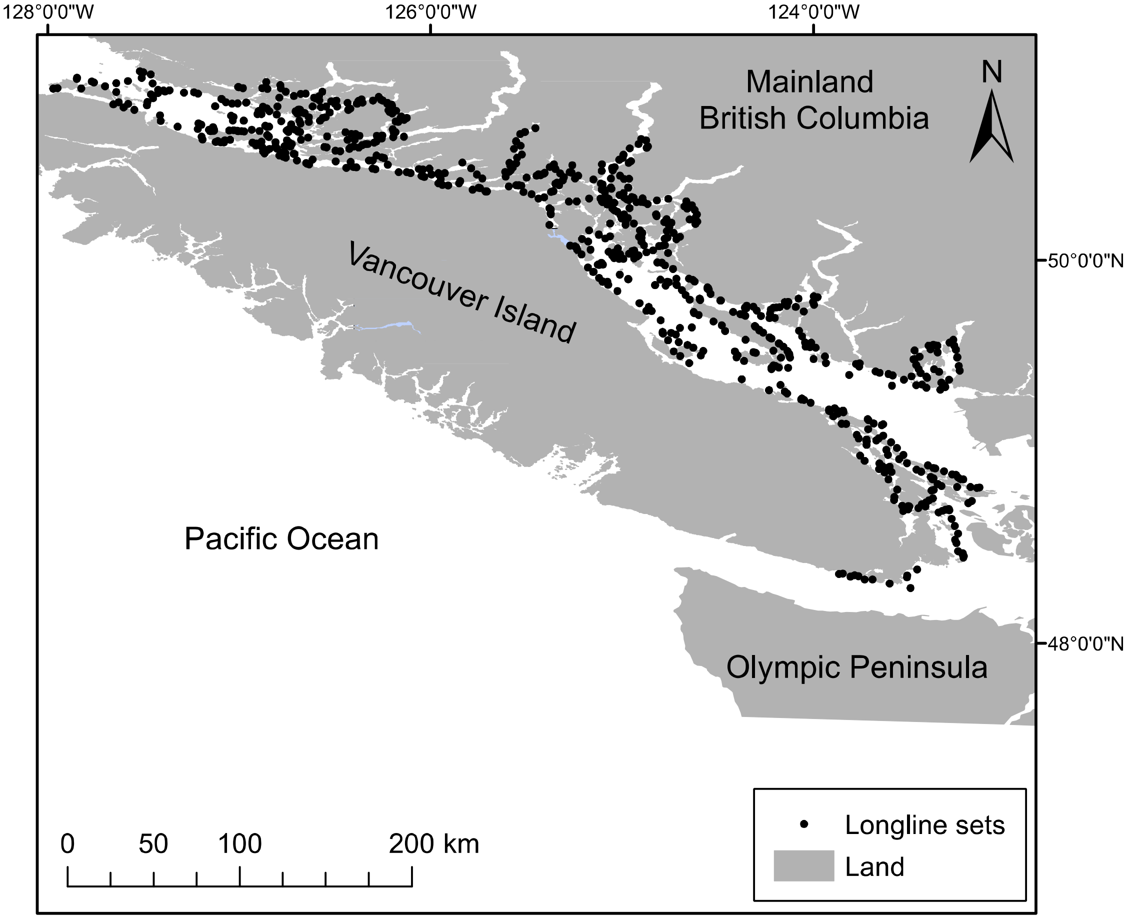


Figure 5.1. Map of the study area depicting all the sets fished from 2003-2015 (excluding 2006).

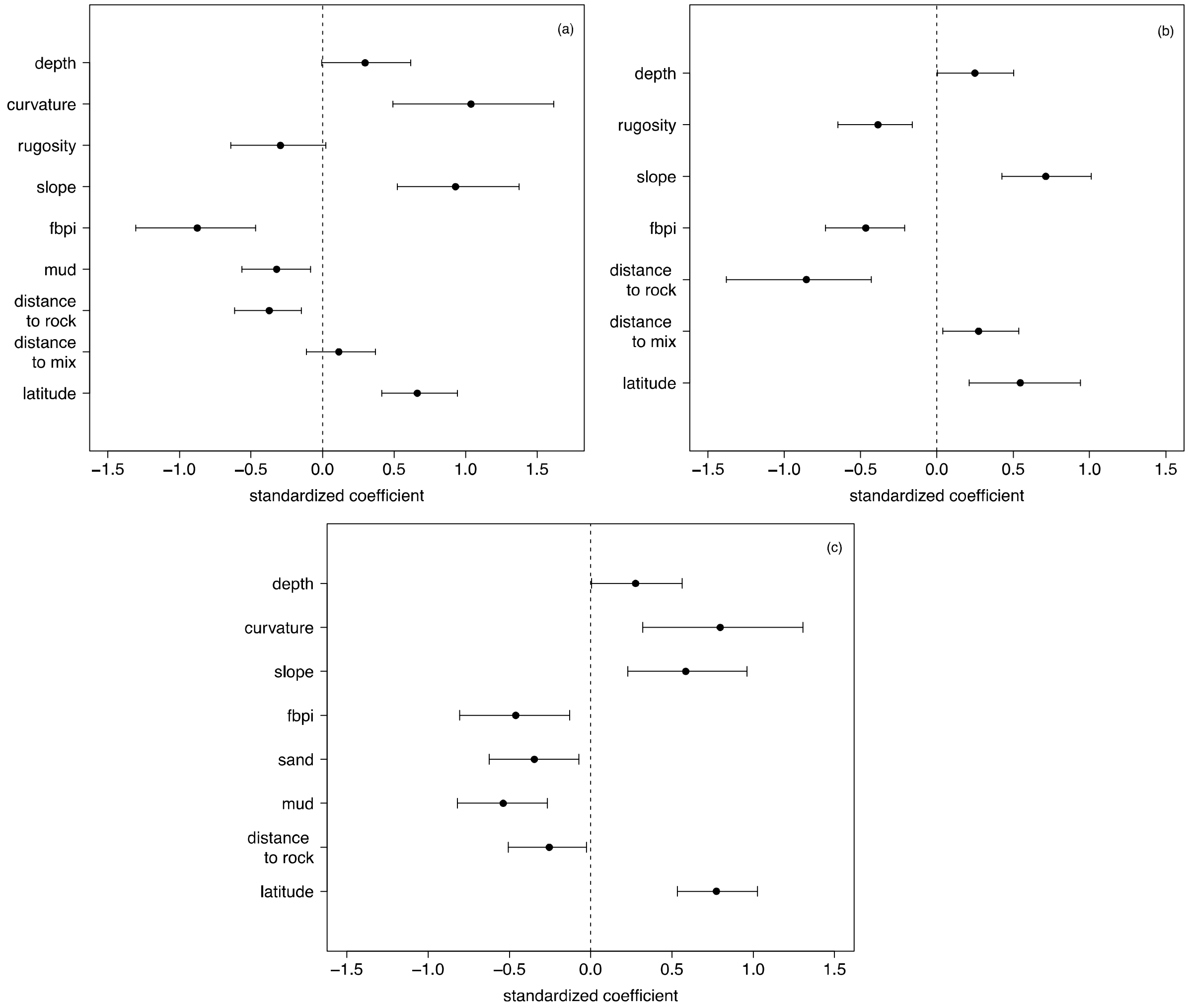


Figure 5.2. Standardized coefficients of the predictor variables retained by the best binomial GLMM model based on AIC scores for a) total rockfish, b) Yelloweye rockfish and c) Quillback rockfish. Horizontal lines represent 95% confidence intervals. Positive values show a positive correlation while negative values represent a negative relationship. Values are significant ($p < 0.05$) if confidence intervals do not overlap with zero (vertical dashed line).

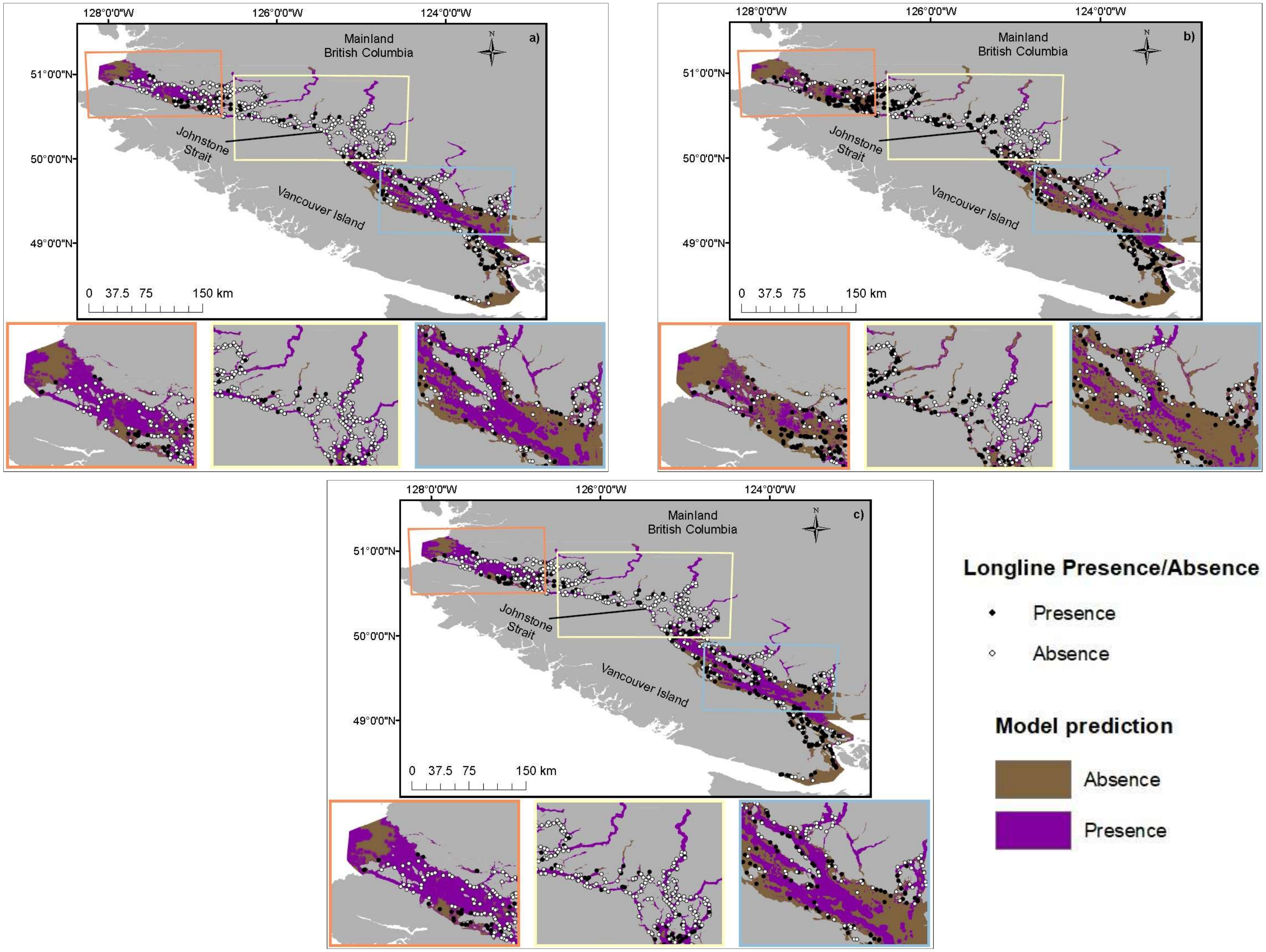

128°0'0"W
126°0'0"W
124°0'0"W
51°0'0"N
50°0'0"N
49°0'0"N
Mainland
British Columbia
Johnstone
Strait
Vancouver Island
0 37.5 75 150 km
N
a)
b)
c)
Longline Presence/Absence
Presence
Absence
Model prediction
Absence
Presence

Figure 5.3. Map of inshore waters of southern British Columbia showing locations where probability of occurrence is low (absence) and high (presence) based on the predictors from the GLM model for a) total rockfish, b) Yelloweye rockfish and c) Quillback rockfish. Insets depict zoomed in areas where black longlines represent absence of rockfish while white longlines represent presence of rockfish. White lines overlaid on purple or black lines overlaid on brown show the agreement between train and test data. In contrast, when white lines overlay a brown area or black lines overlays a purple area the agreement is incorrect (i.e. model prediction is poor).

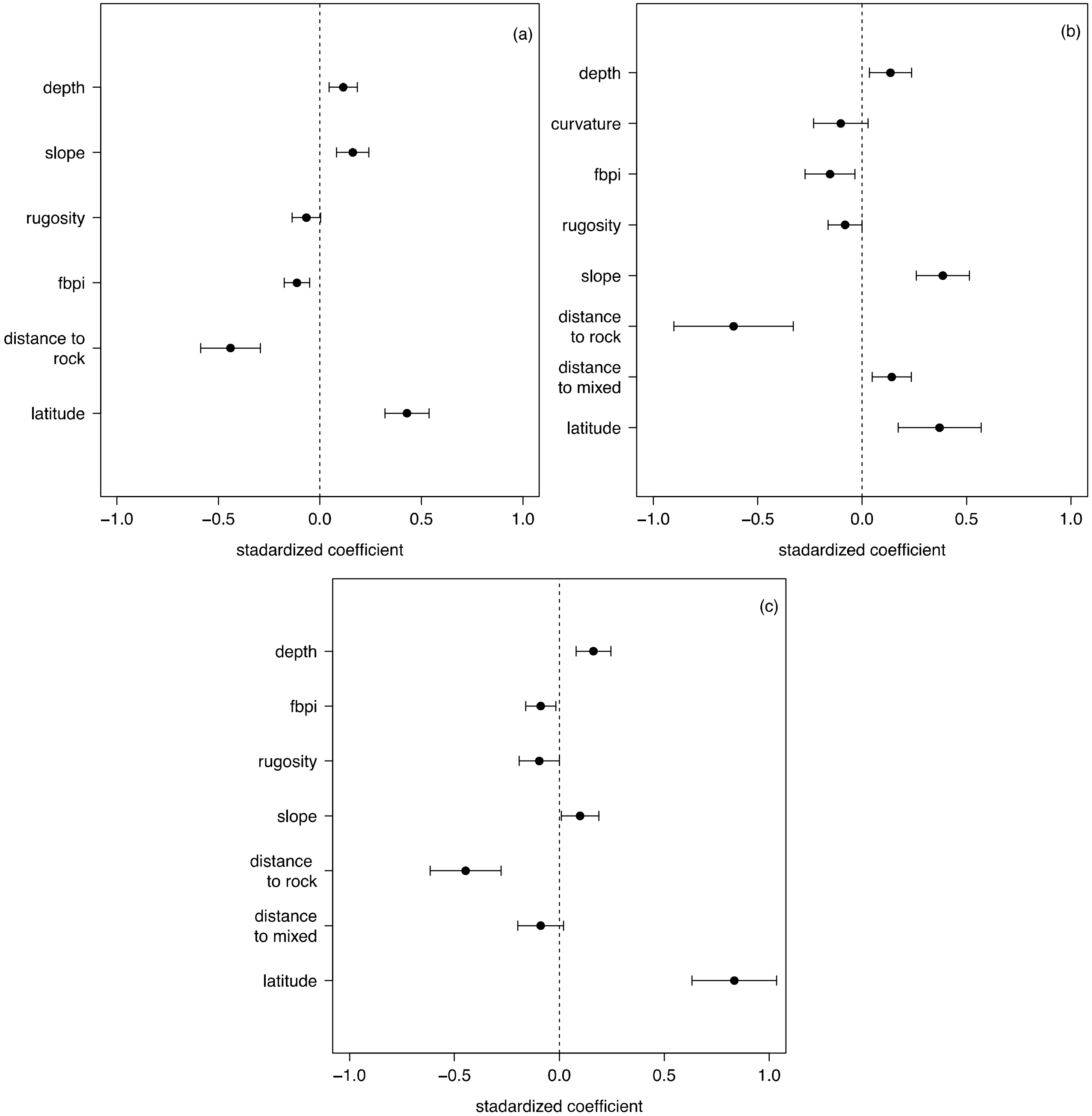
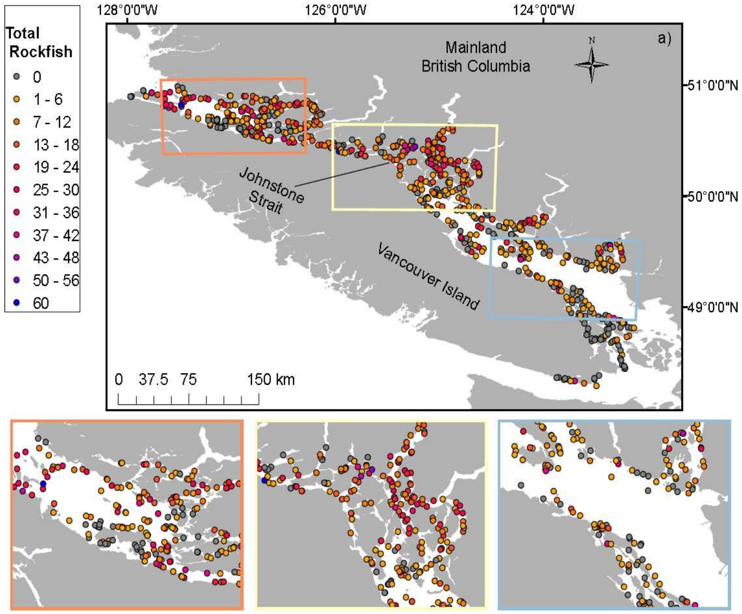
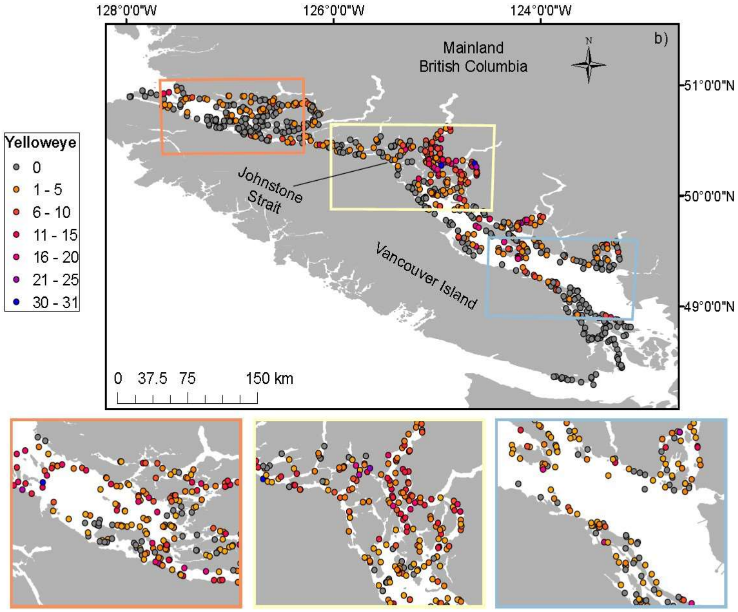

(a)
depth
slope
rugosity
fbpi
distance to rock
latitude
−1.0
−0.5
0.0
0.5
1.0
stadardized coefficient
(b)
depth
curvature
fbpi
rugosity
slope
distance to rock
distance to mixed
latitude
−1.0
−0.5
0.0
0.5
1.0
stadardized coefficient
(c)
depth
fbpi
rugosity
slope
distance to rock
distance to mixed
latitude
−1.0
−0.5
0.0
0.5
1.0
stadardized coefficient

Figure 5.4. Standardized coefficients for the count portion of the best zero-inflated model based on the AIC scores for a) total rockfish, b) Yelloweye rockfish and c) Quillback rockfish. Horizontal lines represent 95% confidence intervals. Positive values show a positive correlation while negative values represent a negative relationship. Values are significant ($p < 0.05$) if confidence intervals do not overlap with 0 (vertical dashed line).


Total Rockfish
0
1 - 6
7 - 12
13 - 18
19 - 24
25 - 30
31 - 36
37 - 42
43 - 48
50 - 56
60
128°0'0"W
126°0'0"W
124°0'0"W
Mainland
British Columbia
N
a)
51°0'0"N
50°0'0"N
49°0'0"N
Johnstone
Strait
Vancouver Island
0
37.5
75
150 km


Yelloweye
0
1 - 5
6 - 10
11 - 15
16 - 20
21 - 25
30 - 31
128°0'0"W
126°0'0"W
124°0'0"W
Mainland
British Columbia
N
b)
51°0'0"N
50°0'0"N
49°0'0"N
Johnstone
Strait
Vancouver Island
0
37.5
75
150 km

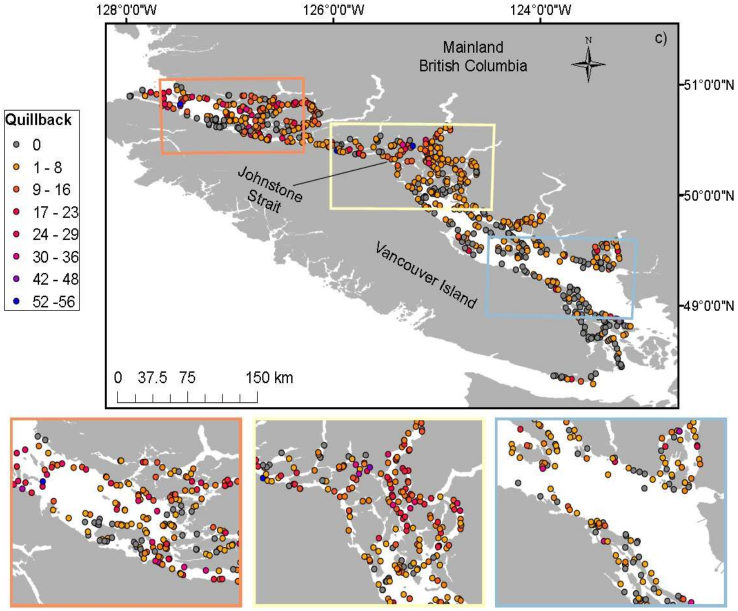

Quillback
0
1 - 8
9 - 16
17 - 23
24 - 29
30 - 36
42 - 48
52 -56
128°0'0"W
126°0'0"W
124°0'0"W
Mainland
British Columbia
N
c)
51°0'0"N
50°0'0"N
49°0'0"N
Johnstone
Strait
Vancouver Island
0
37.5
75
150 km

Figure 5.5. Spatial distribution and abundance of a) total rockfish, b) Yelloweye rockfish and c) Quillback rockfish in inshore waters of southern British Columbia. Insets depict zoomed in areas.

Chapter 6 - Discussion

Marine fish and crustaceans often use different coastal habitats as a function of ontogeny (Nagelkerken et al. 2015). Consequently, it is crucial to understand how such organisms use nearshore habitats (Beck et al. 2001) in order to improve conservation strategies given the multiple threats that coastal ecosystems face (Crain et al. 2008; Crain et al. 2009). In this dissertation, the overall goal was to explore habitat use by fish (and some crustaceans) in mangrove-dominated systems in the tropics and in rocky reefs in temperate waters of BC at global, regional and local scales as a potential mechanism to help sustain small-scale fisheries. In particular, I sought to: i) Determine the mangrove-fishery relationship at a global scale conducting a meta-analysis. ii) Investigate the importance of mangrove habitats in a tropical lagoon system in the Colombian Caribbean. iii) Explore the mangrove-fishery linkage from a Local Ecological Knowledge approach in Ciénaga Grande de Santa Marta; and iv) Relate rockfish habitat characteristics) to performance of species distribution models at large spatial scales (>100 kms) in inshore waters of Vancouver Island, British Columbia. Here, I demonstrate the importance of nearshore habitats in both tropical and temperate systems, for fish and some crustaceans at global, regional and local scales using a suite of different statistical techniques. I was able to demonstrate the importance of mangrove-dominated systems as critical habitats based on the mangrove-fishery linkage (Chapter 2); this habitat relationship has been debated extensively (See for example Lee 2004). Mangrove habitats were also determined to be important habitats in a lagoon system in the Caribbean based on fish sampling (Chapter 3) and using a Local Ecological Knowledge (LEK) approach (Chapter

4) although the mangrove effect was less clear with the fish sampling.

Similarly, rocky reef habitats in inshore waters of British Columbia accurately predict Rockfish abundance and distribution, particularly for Yelloweye and Quillback Rockfish at large spatial extents (100s km). The importance of different habitats and in different systems highlights the importance of improved spatial and conservation management to recover and enhance fish production, particularly important for artisanal fishers whose life styles depend on the resource.

6.1. Effect of mangrove habitats on fisheries

In tropical and subtropical regions, mangrove coverage has consistently been related to higher fishery catches (e.g. Chong 2007). While many policies and conservation plans across the globe were built under this premise, previous reviews had only addressed the relationship from a qualitative perspective, but a rigorous quantitative review was lacking. The meta-analysis presented in Chapter 2 provides strong evidence on the importance of mangrove area in supporting local fisheries in a number of different mangrove settings. The only significant moderator in the analysis was Country, suggesting that at the regional level there can be geomorphological and hydrodynamical settings that may drive the mangrove-fishery linkage (Fig. 2.3). While this chapter contributes to our knowledge by combining all previous studies into one single quantitative analysis, there are some gaps in the available data that must be filled. First, the number of countries where these type of studies have been conducted is low (4.8%) relative to the number of countries that have mangroves in their coast lines (Spalding et al. 2010). As such, addressing this question in countries from Africa and South America (not represented in the study) may provide better insight into the relationship.

Second, having species-specific catch data may offer a more detailed explanation than binning groups. For example, in the Caribbean a species-specific analysis showed that mangrove area correlates with the abundance of eight reef species (Serafy et al. 2015). However, collecting these data is challenging in developing countries because catches in this system are usually multispecies (Rueda et al. 2011) and sampling is not conducted regularly, particularly in small scale fisheries.

One limitation of this study is the metric used to quantify the relationship, the correlation coefficient, because correlation does not necessarily imply causation (Lee 2004). However, there are ecological mechanisms that have been empirically tested across the world to explain this relationship of which the most tested and debated is the nursery hypothesis (Nagelkerken et al. 2015; Sheaves et al. 2015), but others include the importance of mangroves as feeding grounds (Laegdsgaard and Johnson 2001).

In Chapter 3, I explored fish habitat use in a lagoon system in the Caribbean because there had not been any previous studies conducted in such systems in the Caribbean realm. Rather, most studies have been conducted on islands where the seascape differs substantially from lagoon systems (Nagelkerken et al. 2017). Although I found some evidence of mangrove use by fish, I believe that because of the multiple disturbances that the system is presently facing the importance of mangroves as habitat is reduced or hampered. For example, the salinity in some sites was extremely high

(51PSU, Fig. 3.2-3.4) and was the main driver of the metrics observed. While salinity variations are the norm in estuaries and are responsible for the changes in fish assemblages (Sosa-López et al. 2007), these high values in the Ciénaga Grande de Santa Marta (CGSM) are caused by high sedimentation the impedes fresh water from accessing the system (Vilardy et al. 2011). As a consequence, the otherwise regular seasonal variations are altered resulting in changes in abundance and distribution of species.

I sampled using gillnets of different mesh sizes provided by fishers and thus, was limited to part of the total size spectra of the fish community. Such constraints, limited my conclusions regarding whether mangroves are nursery habitats for some species. However, in high turbid waters with micro-tidal regimes sampling for age-0 fish is challenging because visual censuses (diving and stereo video techniques) and catching fish based on tides are ineffective. Thus, future research must focus on creative alternatives to sample the lower end of the spectra to determine if small fish are uniquely using the mangroves prior to ontogenetic shifts (Nagelkerken et al. 2001; Nagelkerken et al. 2002; Nagelkerken et al. 2008) or is it only larger juvenile fish that move in and out of mangroves.

The importance of mangroves to support fisheries has usually been approached from a scientific perspective (e.g. Aburto-Oropeza et al. 2008) without including fishers who are the major stakeholders. Conversely, when including fishers in studies of mangrove-dominated systems seldom is the mangrove-fishery linkage analyzed in depth. Rather, questions about the goods and services are asked to understand how fishers use mangroves. As a novel approach, I combined fishery data and mangrove use data obtained from semi-structured interviews. LEK surveys in the CGSM showed similar patterns in species distribution and catch perception to scientifically collected data. For example, the distribution of three fish species (Fig. 4.5) showed similar patterns compared to an interpolation approach (Rueda 2001). Similarly, catch perception by fishers over the last decade concur with the trends observed by conducting surveys of fisher landings in the same system (INVEMAR 2017). Fishers agreed that mangroves are important for their extracting activity, as they are critical habitats that provide nurseries for juveniles, a food source for some fish and spawning grounds for other fish. Thus, in their opinion, the lack of mangroves would readily decrease populations and limit their catches.

I provide compelling evidence suggesting that fishers' knowledge is accurate compared to scientific knowledge. However, these types of studies have some limitations. For example, the respondents may provide inaccurate data, particularly for previous time periods (Brook and McLachlan 2008; De Freitas and Tagliani 2009) and spatial drawings in order to protect their most favorable fishing grounds (Maurstad 2002; Close and Hall 2006). However, including interviewers that were former colleagues who fishers could trust minimized the problem. Although some authors have suggested that a mapping approach is invasive and can create manager-fisher and fisher-fisher confrontations (Maurstad 2002) when revealing their preferred fishing grounds, I believe that these problems arise in more technological advanced fisheries where travelling is not limited by lack of or small motor power.

6.2. Rockfish habitats in BC

Although species distribution models are common in both terrestrial and marine environments (Elith and Leathwick 2009), a large spatial scale analysis for rockfish had not been conducted. As such, I used previously collected Yelloweye and Quillback data to predict presence-absence and abundance based on derived bathymetry habitat characteristics at a 20m resolution. The results show that Rockfish abundance/presence are related to more complex habitats such as areas with higher slope. More importantly, these findings provide important habitat insight that may be used to reevaluate Rockfish Conservation Areas (Yamanaka and Logan 2010) as one solution to continue efforts to rebuild rockfish stocks in inshore waters of Vancouver Island (Yamanaka et al. 2006a; Yamanaka et al. 2006b).

Although medium resolution data (20m) accurately described some important and previously observed habitat characteristics used by rockfish (Young et al. 2010; Young and Carr 2015) as higher resolution data become available in the Strait of Georgia, Johnston Strait, and Queen Charlotte Strait (e.g. 5m) predictions will increase in accuracy and habitat characteristics at finer scale will emerge. Using longline surveys to predict rockfish proved to be a reasonable methodology to conduct species distribution models; however, some finer scale information is lost by averaging data across the length of the line. As consequence, ROVs that capture presence-absence of fishes at the same scale as the available bathymetry data will improve the accuracy of the models.

6.3. The importance of habitat for fisheries

In an Ecosystem-Based Fishery Management (EBFM) approach all ecological processes acting in one given ecosystem are taken into consideration to obtain healthy fisheries over time while maintaining a healthy ecosystem (Pikitch et al. 2004). Thus, one of the key components of this approach is preserving marine habitats where many ecological processes occur. In this dissertation, I provide evidence supporting the idea that nearshore habitats play a fundamental role to support artisanal fisheries in both tropical and temperate systems. Mangroves, for example, are critical fish and crustacean habitats that enhance fisheries (Chapter 2 and Chapter 4) based on statistical and LEK approaches. Similarly, more complex rocky reefs better predict rockfish abundance and presence-absence.

Although the presence of coastal habitats contributes to maintain fisheries, the production of the resource may be limited in the presence of these habitats if other variables of the ecosystem are beyond tolerance limits. For example, despite the presence of mangroves in some areas of the CGSM I found that the major driver of abundance and diversity was salinity. High salinities in the system are a consequence of previous human impacts whereby freshwater channels that feed fresh water to the system were interrupted. When reopened there were high levels of sedimentation that decreased freshwater flow, and, consequently, produced higher salinity (Botero and Salzwedel 1999; Vilardy et al. 2011). Furthermore, the high fishing pressure being applied to the system has shown evidence of growth overfishing that may have negative repercussions on species interactions. Similarly, the presence of suitable habitat for rockfish does not guarantee increase in population size if the stocks are overfished. Fortunately, spatial closures and catch regulations have been put in place in order to recover this populations (Eckert et al. 2017).

While habitat conservation plans and restoration programs are an important step forward to maintain sustainable fisheries and healthy ecosystems, other pervasive impacts such as overfishing and water quality must also be included in the process for the ecosystem to recover. Moreover, including fishers as the major stakeholders in such ecosystem-based management plans increases the chances of the management plans being successful (Ban et al. 2013). While ecosystem-based management has proven to be an adequate approach thanks to highly resilient ecosystems (Barnett and Baskett 2015), these approaches are still rare.

6.4. Conclusion

Many people depend on marine resources as a source of food to alleviate poverty (Béné 2006) and cultural reasons (Eckert et al. 2017). However, these resources are being depleted at high rates due to habitat deterioration, overexploitation and other human impacts (Crain et al. 2008; Crain et al. 2009) greatly affecting fishers. This thesis stresses the importance of coastal habitats (mangroves and rocky reefs) to maintain juvenile and adult populations of fish and some crustaceans because a higher structural complexity offers refuge from predators and food. Thus, conservation plans to stop the loss of these systems will have a positive impact on fish and crustacean populations. Nonetheless, in disturbed systems, many other aspects of the ecosystem such as overexploitation and abiotic variables must also be controlled in order to observe a complete recovery of the system. As such, ecosystem based management approaches must be considered to rebuild fish stocks that are essential for small-scale fishers.

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Appendices

Appendix A: Supplemental information for Chapter 3

Dissolved oxygen, pH and temperature did not differ across sites or within habitats ($p > 0.05$) and thus, were discarded from future analysis. In contrast, salinity was similar in four sites (Cen, Cgd, Rin and Clr) but different in Agn (lowest) and Lun (highest). A similar pattern was obtained for turbidity, where a higher turbidity (lower Secchi Disk visibility) was observed in Agn (13.3cm) and Lun (19.33cm) while lower turbidity was obtained at the remaining four sites (Table A3.1).

Table A3.1. Mean (± SD) of the environmental variables that differed only across sites or that had no statistical difference across sites. Cen = Centro; Cgd = Caño Grande; Rin =Rinconada; Clr = Ciénaga la Redonda; Agn = Aguas Negras; Lun = Luna. ** p_values < 0.001; * p_value <0.05

Site	**Salinity (PSU)**	**DO (mg/L)**	**Temp (°C)**	**Turbidity (cm)**	**pH**
Cen	38.40 (0.47)	6.35 (1.31)	31.83 (1.10)	33.61 (7.03)	8.61 (0.21)
Cgd	36.70 (0.66)	6.96 (1.32)	31.78 (1.10)	30.00 (6.64)	8.71 (0.21)
Rin	36.41 (0.95)	6.59 (1.53)	30.82 (0.89)	30.55 (7.45)	8.75 (0.22)
Clr	37.73 (5.62)	7.1 (1.46)	31.44 (0.95)	28.33 (5.69)	8.5 (0.16)
Agn	8.55 (2.88)**	6.86 (1.07)	32.36 (1.43)	13.33 (3.62)**	8.88 (0.24)
Lun	51.88 (5.42)**	6.8 (1.38)	32.77 (0.99)	19.33 (1.76)**	8.42 (0.13)

Table A3.2. Fish abundance model results based Akaike's Information Criterion (AIC). Best model is showed first followed by all other competing models. Larger ΔAICc (i.e. ΔAIC > 2) and lower weight represent less suitable models. All models were fitted with site as a random effect.

model	df	AICc	ΔAIC	weight
salinity + habitat	7	721.54	0	0.28
salinity + distance + habitat	8	722.54	0.99	0.17
salinity * habitat	10	723.52	1.98	0.11
salinity + turbidity + distance + habitat	9	723.59	2.05	0.10
salinity + turbidity + mangrove density (water) + distance + habitat	10	725.80	4.25	0.03
salinity + turbidity	5	725.91	4.36	0.03
mangrove density + mangrove density (water) + distance	6	727.58	6.04	0.01
salinity + turbidity + mangrove density + mangrove density (water) + distance + habitat	11	727.71	6.17	0.01

Table A3.3. Fish biomass model results based on Akaike's Information Criterion (AIC). Best model is showed first followed by all other competing models. Larger ΔAICc (i.e. ΔAICc > 2) and lower weight represent less suitable models. All models were fitted with site as a random effect.

model	**df**	**AICc**	**ΔAIC**	**weight**
salinity + turbidity + distance + mangrove density (water) + habitat	10	1667.91	0	0.15
salinity + turbidity + distance + habitat	9	1668.2	0.15	0.14
salinity + turbidity + distance + mangrove density + mangrove density (water) + habitat	11	1668.33	0.38	0.12
salinity + turbidity + distance + broad habitat	7	1668.78	0.83	0.1
salinity + turbidity + habitat	8	1668.98	1.04	0.09
salinity + turbidity + distance + mangrove density (water) + habitat	8	1669.01	1.07	0.09
salinity + habitat	7	1669.39	1.44	0.07
salinity + turbidity + mangrove density (water) + habitat	9	1669.40	1.46	0.07
Null	3	1669.51	1.56	0.07
salinity + turbidity + distance + mangrove density + mangrove density (water) + broad habitat	9	1670.09	2.15	0.05
distance + mangrove density + mangrove density (water) + habitat	9	1670.99	3.043	0.03
salinity + turbidity + mangrove density (water) + broad habitat	7	1672.99	5.05	0.012

Table A3.4. Fish diversity model results based on Akaike's Information Criterion (AIC). Best model is shown first followed by all other competing models. Larger ΔAICc (i.e. ΔAICc > 2) and lower weight represent less suitable models. All models were fitted with site as a random effects except for the last model that was fitted with habitats nested within sites as a random effects (habitat|site)

model	df	AICc	ΔAIC	weight
salinity + turbidity + distance + habitat	9	136.85	0.00	0.34
salinity + turbidity + habitat	8	137.20	0.35	0.29
salinity + turbidity + mangrove density (water) + distance + habitat	10	138.43	1.58	0.15
salinity + turbidity + mangrove density (water) + mangrove density + distance	8	138.71	1.86	0.14
salinity + turbidity + mangrove density (water) + mangrove density + distance + habitat	11	140.95	4.10	0.04
salinity + turbidity + mangrove density (water) + mangrove density + habitat	10	141.37	4.52	0.04
mangrove density (water) + mangrove density + distance + habitat	9	145.30	8.45	0.01
salinity + turbidity + mangrove density (water) + mangrove density + distance + habitat	20	163.47	26.62	0.00

Table A3.5. Maturity model results based on Akaike's Information Criterion (AIC). Best model is shown first followed by all other competing models. Larger ΔAICc (i.e. QAICc > 2) and lower weight represent less suitable models. All models were fitted with site + observation as random effects except for the last model which was only fitted with site as a random effect.

model	df	AICc	ΔAIC	weight
habitat+ salinity + turbidity	8	426.46	0	0.45
distance + habitat+ salinity	9	427.34	0.88	0.29
distance + habitat+ salinity + turbidity + mangrove density (water)	10	428.92	2.47	0.13
distance + habitat+ mangrove density (water) + mangrove density	9	429.94	3.48	0.08
distance + habitat+ salinity + turbidity + mangrove density (water) + mangrove density	11	430.52	4.07	0.06
distance + habitat+ salinity + turbidity + mangrove density (water) + mangrove density	10	489.3	62.84	0

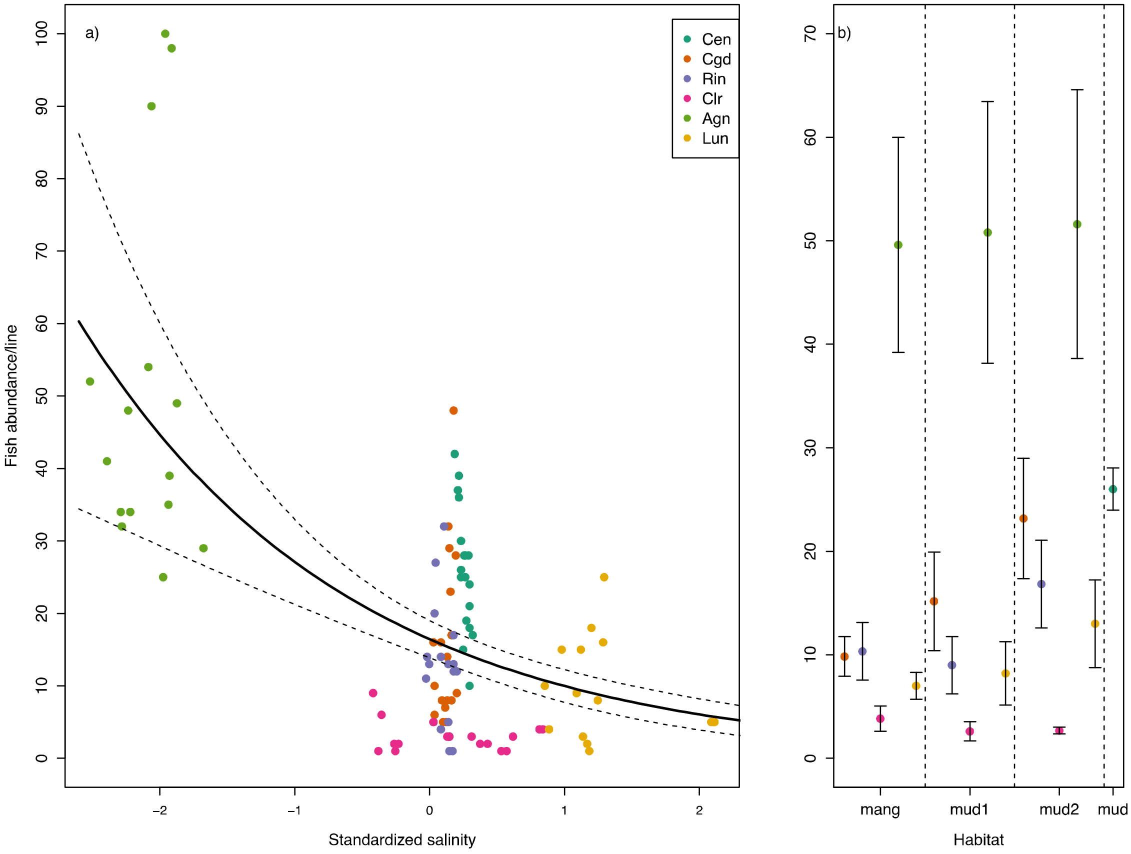

Figure A3.1. Fish abundance relationship with a) salinity and b) habitats based on the best GLMM evaluated by AIC$_c$. Salinity values are standardized depicting site variation. In panel a black line represents the fitted model and doted lines represent 95% confidence intervals. In panel b, points represent mean fish abundance estimated from the model and error bars are the standard errors (sem). Habitats are: mang = mangrove habitat; mud1 = mud habitat further from mangroves; mud2 = mud habitat furthest from mangroves and mud = habitat in site Cen.

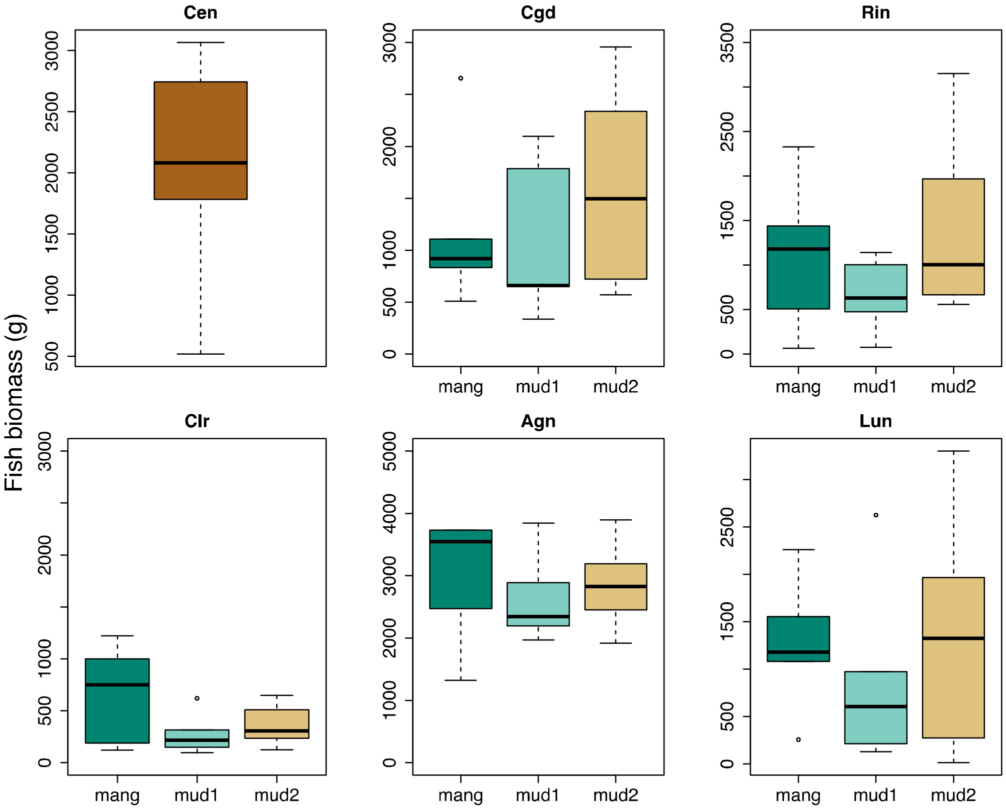

Figure A3.2. Boxplots for each site showing the variability of fish biomass across habitats. Cen had one habitat, mud. The other habitats are: mang = mangrove habitat; mud1 = habitat further from mangroves and mud2 = habitat furthest from mangroves. Sites are: Cen = Centro, Cgd = Caño Grande, Rin = Rinconada, Clr = Cienaga La redonda, Agn = Aguas negras and Lun = Luna.

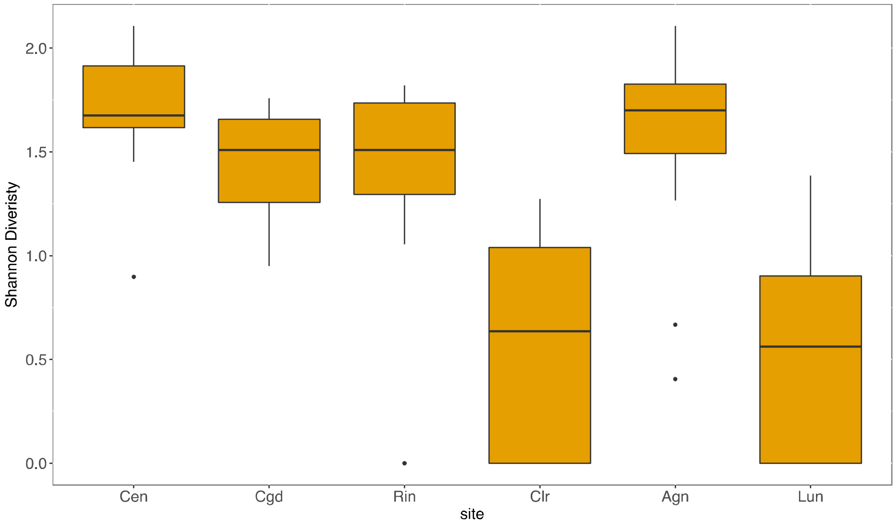

Figure A3.3. Fish diversity boxplots evaluated by the Shannon index at each site showing the variability within site and differences across sites. Sites are: Cen = Centro, Cgd = Caño Grande, Rin = Rinconada, Clr = Cienaga La redonda, Agn = Aguas negras and Lun = Luna.

Appendix B: Supplemental information for Chapter 4

Effect of the mangrove forest in the artisanal fishing in
Ciénaga Grande de Santa Marta

Date:

Participant’s code:

Interviewer:

1. Fishing

a. When did you start fishing in the lagoon system?

b. What fishing gear have you employed the most in the past year?

c. Has the fishing gear changed since you started fishing in the lagoon system?

YES_____ No_____

   i. If you answered YES, what gear did you use before the one currently used?

   ii. In what year did you change the preferred fishing gear.

d. Normally, how many times per week do you fish?
e. How many times per week?
   i. In the dry season

   ii. In the rainy season

f. In the map, indicate the preferred fishing site.

g. In the map, indicate the region where you normally fish.

   i. In the dry season

ii. in the rainy season

h. Indicate the grid where you have fished lately.

i. In the indicated region, draw the most visited fishing area within the last 12 months in terms of:

i. Preferred species

ii. Best catch?

j. Indicate the grid where you fished 5 years ago.

k. In the indicated region, draw the most visited fishing area 5 years ago, in terms of:

i. Preferred species

ii. Best catch?

l. Indicate the grid where you fished 10 years ago.

m. In the indicated region, draw the most visited fishing area 10 years ago, in terms of:

i. Preferred species

ii. Best catch?

n. Complete the following table:

	Current year	**5 years ago**	**10 years ago**
Which were the five most fished species.	**1.** **2.** **3.**	**1.** **2.** **3.**	**1.** **2.** **3.**

	4. 5.	4. 5.	4. 5.
From the five species, has their catch: improved, decreased or stayed the same	1. 2. 3. 4. 5.	1. 2. 3. 4. 5.	1. 2. 3. 4. 5.
From the five species, has their size: improved, decreased or maintained	1. 2. 3. 4.	1. 2. 3. 4.	1. 2. 3. 4.

	5.	**5.**	**5.**

o. How much of your income depends on your fishing activity

p. How many times per week do you consume catch from the lagoon system?

2. Mangroves

a. Answer **Yes** or **No** to the following questions:

i. Are Mangroves plants?

YES___ NO___

ii. Is there more than one mangrove species in the lagoon system?

YES___ NO___

iii. Does the term “mangrove”, include all the mangrove species found in the lagoon system?

YES___ NO___

iv. Are mangroves important habitats for fish?

YES___ NO___

v. Does mangrove loss concern you?

YES___ NO___

vi. If the mangrove is lost, will there be fewer fish in the lagoon system?

YES___ NO___

b. How have you used mangroves in the past?

c. Would you be able to continue extracting such services in the absence of mangroves?

d. Do you think the mangroves are in good or bad condition? Why?

e. How would you describe a mangrove is in good condition?

f. In your opinion, in the last 5 years, mangrove coverage has increased, decreased or stayed the same?

g. In your opinion, in the last 10 years mangrove coverage has increased, decreased or stayed the same?

3. Mangroves and Fisheries

a. Do you think there is a relationship between mangroves and your catch?

YES____ NO____

i. If you answer was YES in the previous question, how do you think they are related?

ii. What do you think would happen to the 5 species that you frequently catch if the mangrove was completely removed?

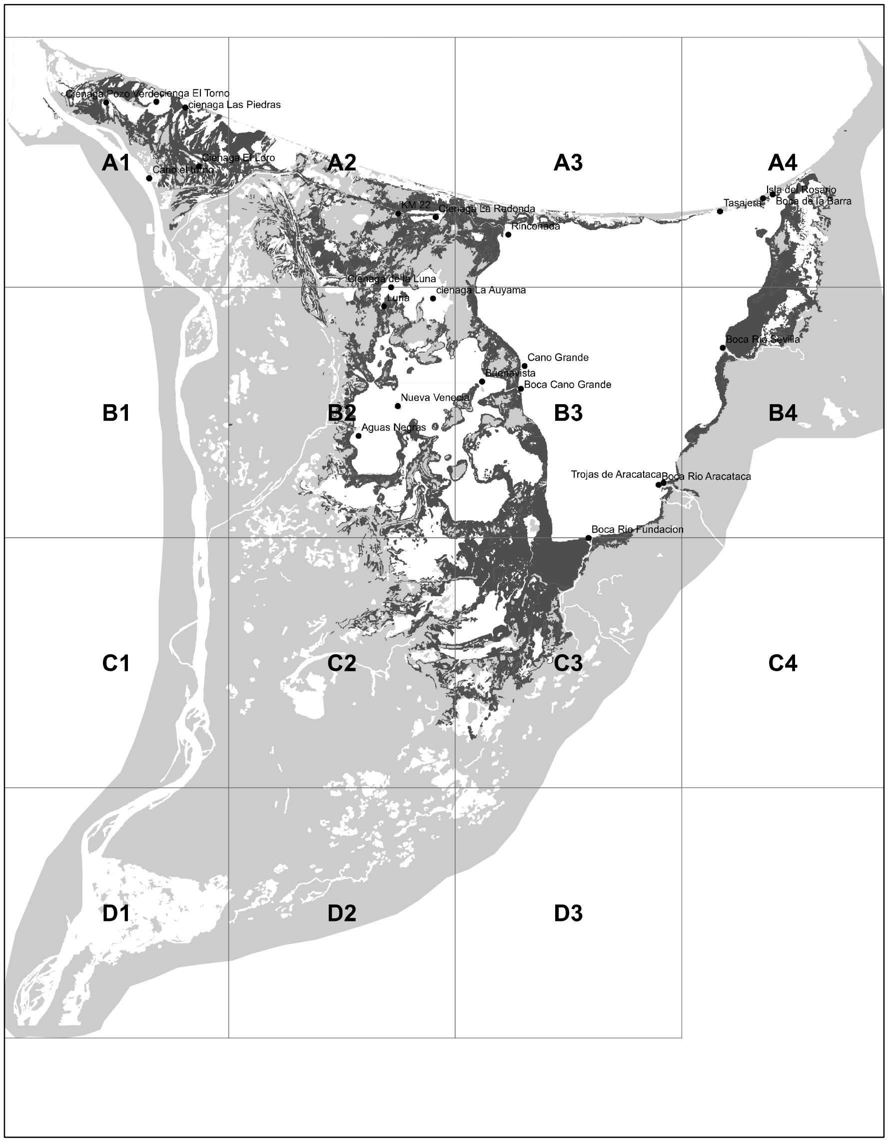

Figure A4.1. Map of Ciénaga Grande de Santa Marta with a 4 x 4 grid to help the fishermen precisely locate their preferred fishing areas.

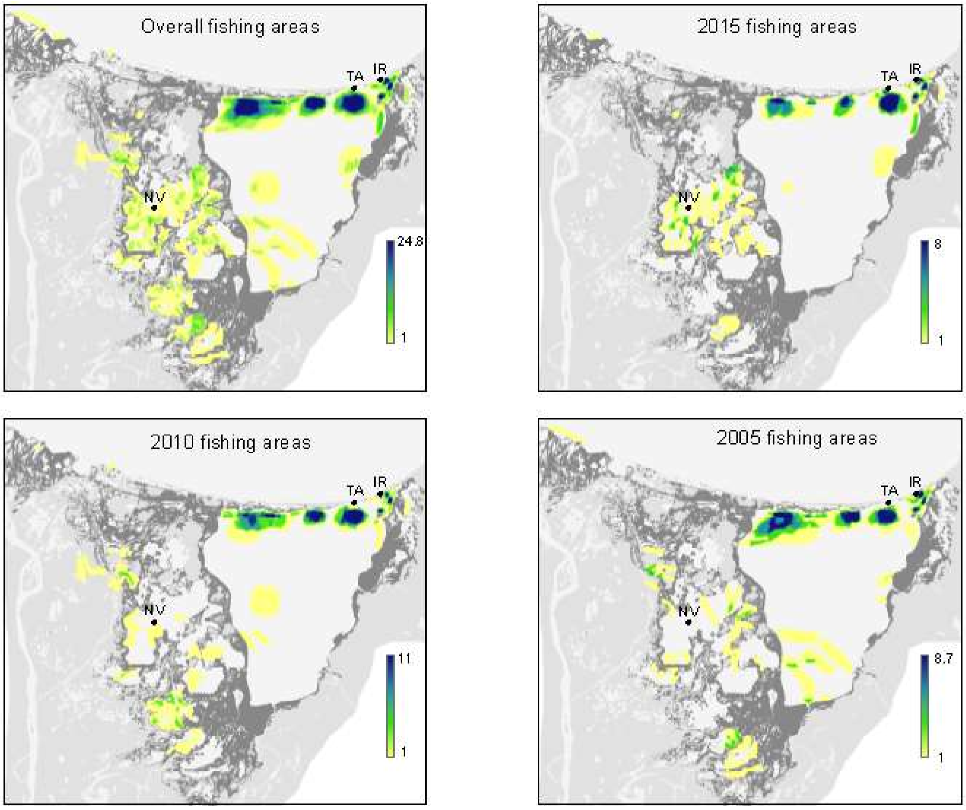

Figure A4.2. Maps showing overall fishing areas over the three time periods, recent (2015), 5 years ago (2010) and 10 years ago (2005). In the legend, yellow represents low polygon overlap while blue represent high polygon overlap and thus higher effort in such areas. Each color ramp differs in scale as the polygon overlap combination differed for each map.

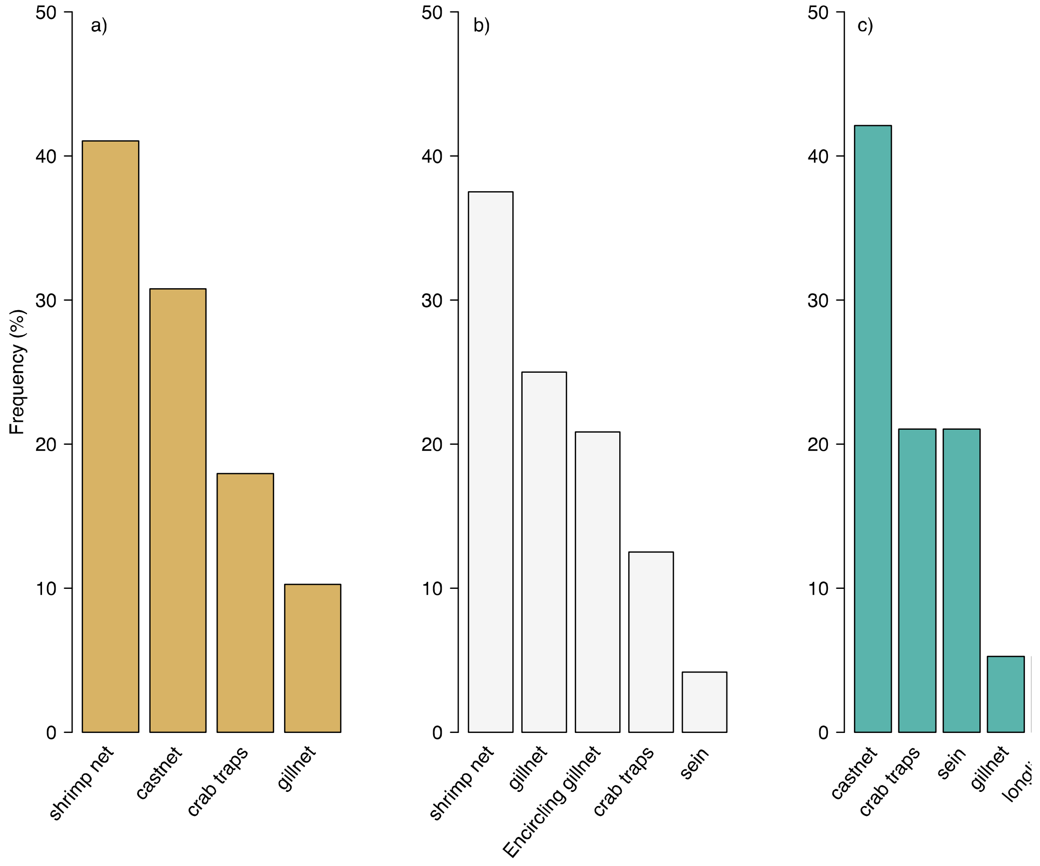

Figure A4.3. Frequency (in percentage) of fishing gear used in a) Isla Rosario (IR); b) Tasajera (TA) and c) Nueva Venecia (NV).

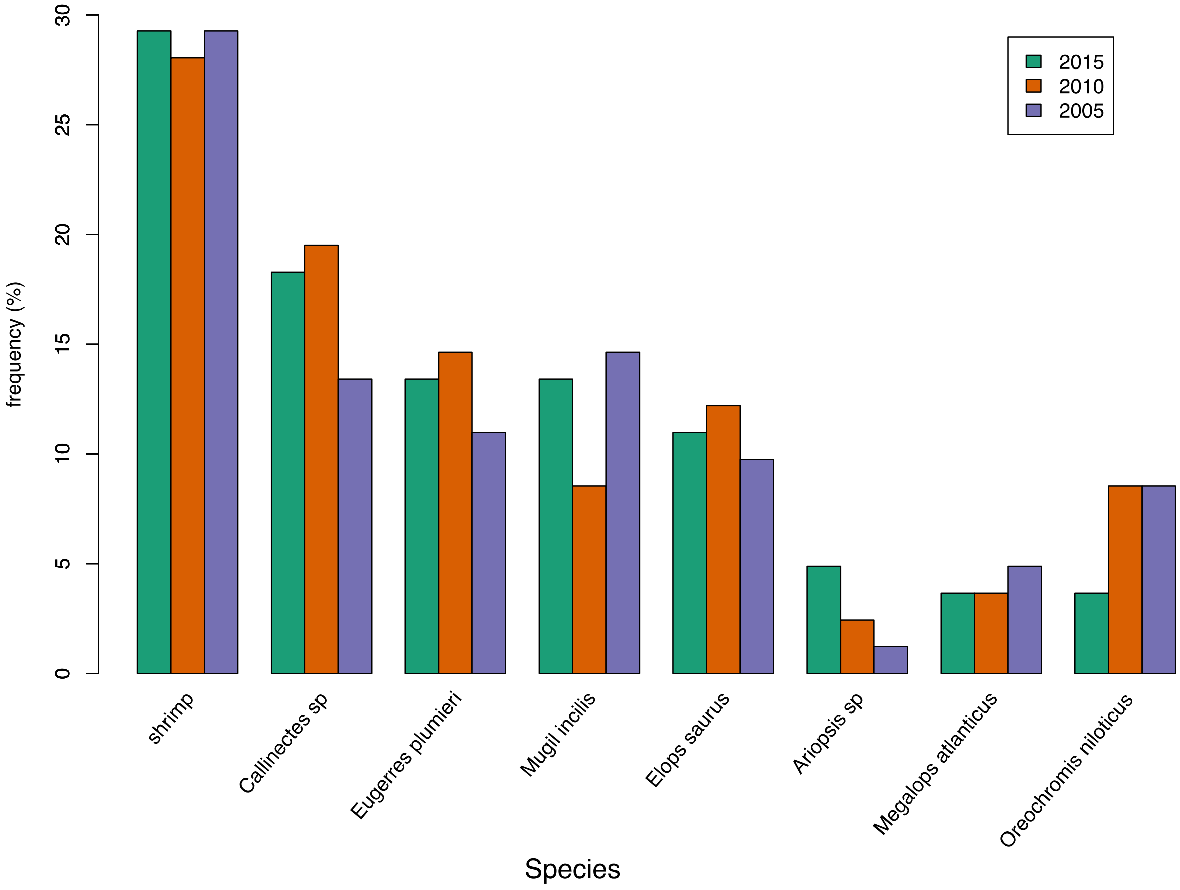

Figure A4.4. Barplot showing frequency (percentage) of the most caught species over time according to the fishers in Ciénaga Grande de Santa Marta.

Appendix C: Supplemental information for Chapter 5

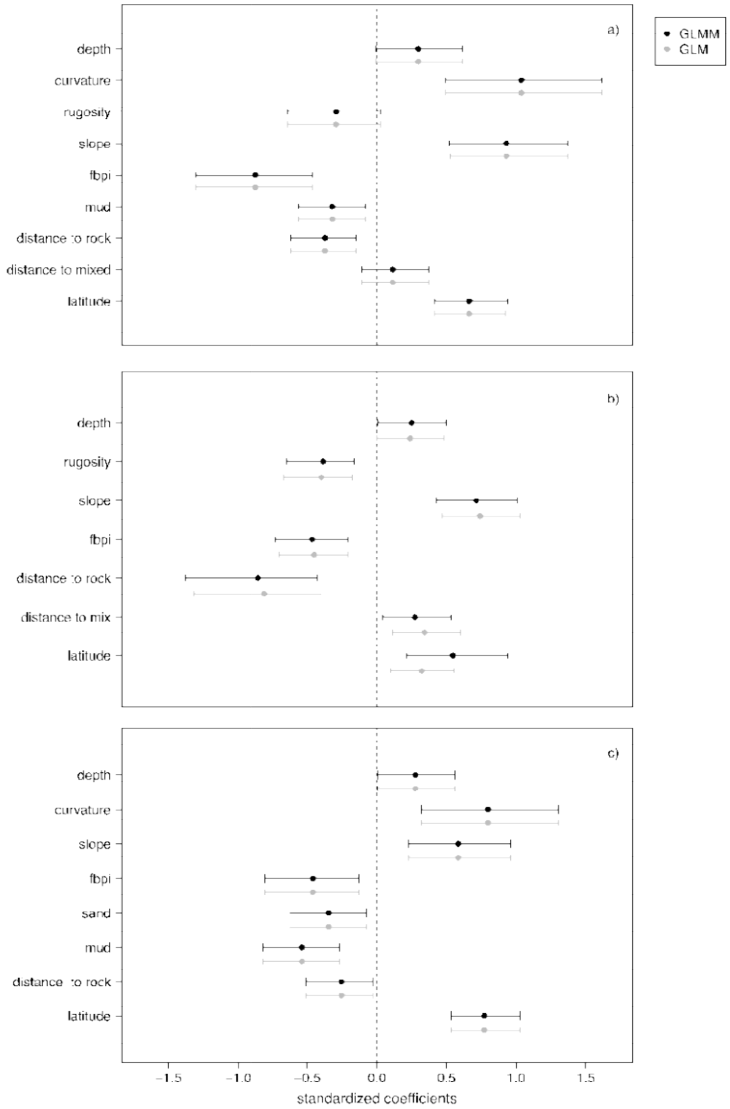

Figure A5.1. Standardized coefficients to predict probability of occurrence using both GLM and GLMM approaches for a) total rockfish, b) Yelloweye rockfish and c) Quillback rockfish from inshore waters of southern British Columbia. Horizontal lines represent 95% confidence intervals. Positive values show a positive correlation while negative values represent a negative relationship. Values are significant ($p < 0.05$) if confidence intervals do not overlap with 0 (vertical dashed line).

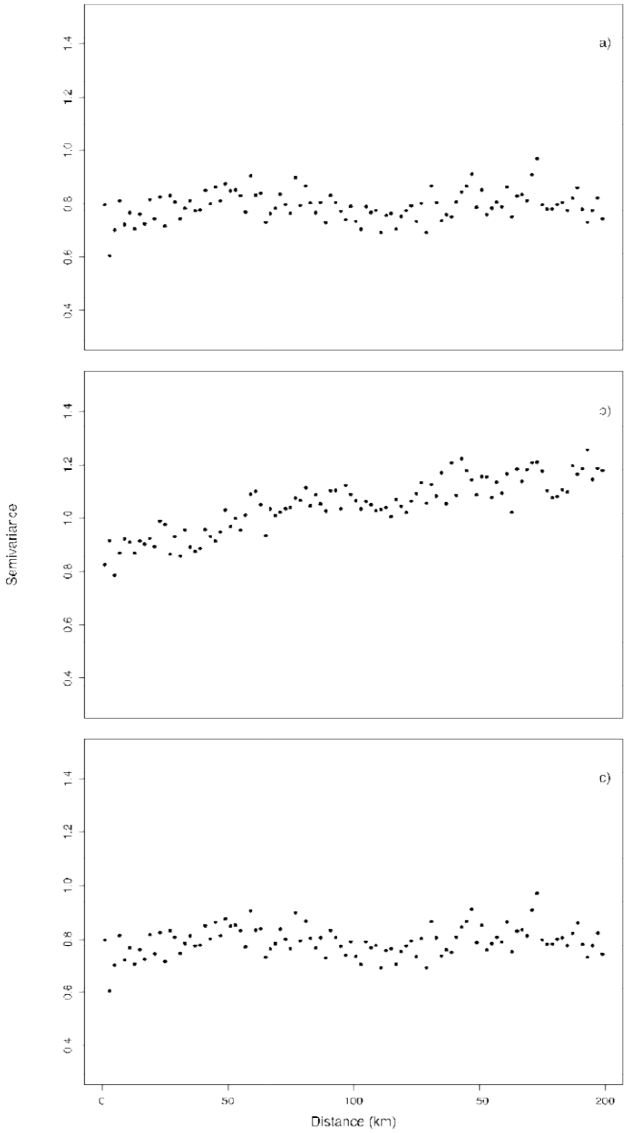

Figure A5.2. Semivariogram to detect spatial autocorrelation for the presence-absence models of a) total rockfish, b) Yelloweye rockfish and c) Quillback rockfish in inshore waters of southern British Columbia.

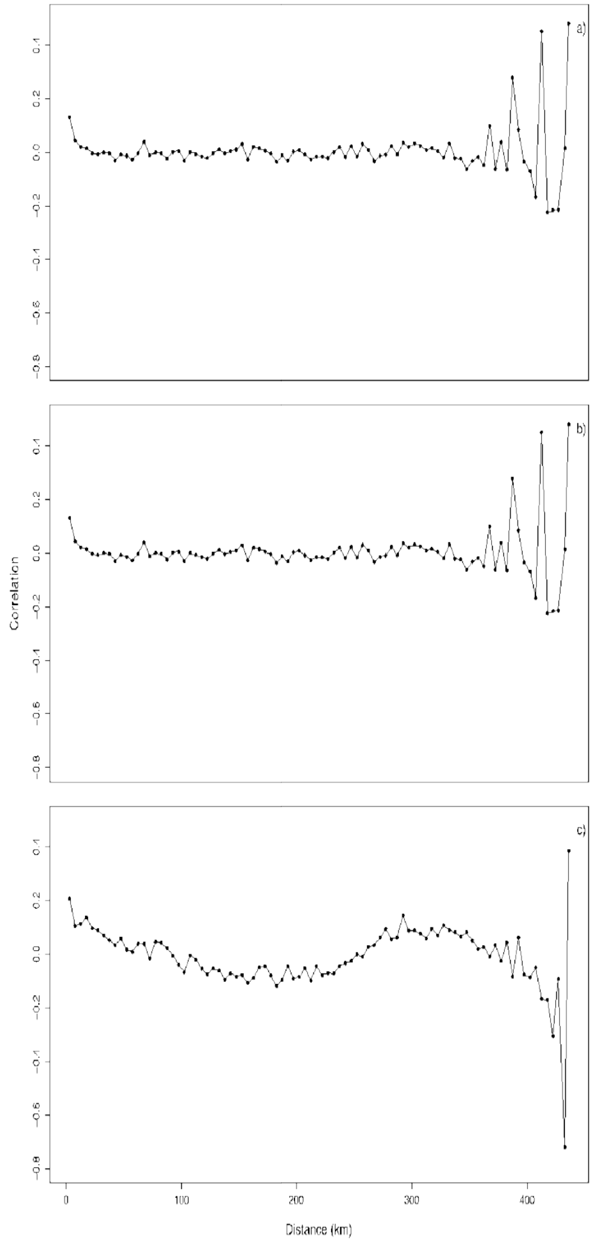

Figure A5.3. Moran I spatial correllograms showing the relationship in abundance of a) total rockfish, b) Yelloweye rockfish and c) Quillback rockfish in inshore waters of southern British Columbia.