SEA TURTLES AND PAPER PARKS IN A NICARAGUAN SMALL SCALE FISHERY

by

Allison Cutting

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The following individuals certify that they have read, and recommend to the Faculty of Graduate and Postdoctoral Studies for acceptance, a thesis entitled:

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Examining Committee:

Terre Satterfield, Professor, Resources, Environment & Sustainability, UBC
Co-supervisor

Rashid Sumaila, Professor, Oceans and Fisheries, Public Policy and Global Affairs, UBC
Co-supervisor

Kai Chan, Professor, Resources, Environment & Sustainability, UBC
Additional Examiner

Abstract

The olive ridley sea turtle (*Lepidochelys olivacea*) experienced a global population decline of about 30% over two generations. Threats include direct take of turtle eggs for human consumption and indirect take of adults via incidental catch in fishing gear, known as bycatch. Chacocente beach in Nicaragua is one of ten mass nesting sites for olive ridleys in the world. The shore of Chacocente is under military protection and the surrounding waters are part of an established marine protected area (MPA). In the neighboring community of El Astillero, 60% of households are economically dependent on fishing. Fishers expressed concern for turtle bycatch in gillnets and managers expressed concern for uncontrolled fishing; however, bycatch in this region has not been quantified. This study examines bycatch in reference to the protected area's design logics to understand whether management strategies have encouraged the desired fisher behavior and outcomes as concerns olive ridleys. Through document analysis, we derived four unintentional assumptions that seem to inform management design. We then compared these unintentional assumptions to 586 net set observations conducted in 2019. Results showed 24 turtles were caught inside the MPA, 24 turtles caught outside the MPA, with 0.04 and 0.03 turtles caught per 100 meters of net, respectively. While bycatch is similar inside and outside the MPA, revealing a lack of compliance with MPA boundaries, bycatch varied greatly between targeted species and month, with relatively high proportions of bycatch occurring in the snook fishery and in September. Such variance is not accounted for in regulations. To build quality protection in practice – not quantity of parks on paper – we aim to improve understanding of fishing interactions at-sea and make recommendations for management design that work for both sea turtle populations and fisher livelihood security.

Lay Summary

Fishing contributes greatly to human food and livelihood security, while also posing major threats to marine species. Marine protected areas are a popular management strategy, which restrict fishing activity to provide refuge to marine life; however, over half of protected areas lack enforcement and do not meet objectives in practice. My research investigates a marine protected area in Nicaragua, created to safeguard olive ridley sea turtles. I tested if regulations encouraged fishers to alter behavior, and if this facilitated turtle protection as expected. I found an equal number of turtles were caught inside and outside the protected area, and that most turtles were caught in September and while fishing for a certain fish, called snook. As such, it appears regulations have not produced desired outcomes and there are opportunities for management to be better tailored to meet needs of fishers and sea turtles, to move towards ecological and economic sustainability.

Preface

This thesis contains my own original work, which has been shaped by collaboration with community members of El Astillero, with members from the non-governmental organization Casa Congo, and with others at the University of British Columbia. Chapter three is intended for publication in a peer review journal. I created the research design, with feedback on the observer program provided by Alejandro Cotto, Gadea Velkiss, Larry Crowder, Elena Finkbeiner, Jose Urtega, John Bieraugel, John Wang, Jody Van Niekerk, Luca Marsaglia, and Manuel Cortez. Luca Marsaglia, Manuel Cortez, and I trained Mirna Berroteron, Joel Palacios, and Jeffery Sandoval in data collection as fisheries observers. They then collected all the quantitative data for this study in El Astillero, Nicaragua. The research frame was developed with guidance from Terre Satterfield and Rashid Sumaila. Luca Marsaglia and I co-created figure 5. He coded the figure and provided feedback on spatial analysis. I conducted all quantitative and qualitative data analysis and writing with iterative feedback from Terre Satterfield and Rashid Sumaila. The majority of analysis and writing took place on the unceded and traditional lands of the Musqueam, Squamish, and Tsleil Waututh peoples, and the traditional lands of the S'Klallam peoples.

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List of Abbreviations

BPUE – Bycatch per unit effort

CITES – Convention on International Trade in Endangered Species of Wild Fauna and Flora

cm – Centimeters

CPUE – Catch per unit effort

FAO – Food and Agriculture Organization

FFI – Fauna and Flora International

FSLN – Sandinista National Liberation Front

GDP – Gross domestic product

INPESCA – Nicaraguan Institute of Fisheries and Aquaculture

IUCN – International Union for Conservation of Nature

Lbs. – Pounds

LED – Light-emitting diode

m – Meters

MARENA – Nicaraguan Ministry for Environment and Natural Resources

MPA – Marine protected area

NGO – Non-governmental organization

NOAA – National Oceanic and Atmospheric Administration

SINAP – National System of Protected Areas

UBC – University of British Columbia

USD – United States Dollar

VPUE – Value per unit effort

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Thank you. These two words feel too simple to accurately convey my gratitude to those who have ventured alongside me in this great learning journey. Yet part of this learning has regarded recognizing what is enough, and so it is fitting to utilize humble words and this small space to communicate appreciation. I trust that the depth of such will be received.

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In memory of Marlon Jose Cruz Morales.

In hope for Allison Isabella Cruz.

Chapter 1: Introduction

Small scale fisheries are critical for livelihood and food security (FAO, 2015). An estimated 260 million people are employed within the marine fisheries sector, and of these, 78 percent live in developing countries (Teh & Sumaila, 2013). Furthermore, small scale fisheries produce 50 percent of the total catch in developing countries, and 95 percent of these landings are consumed locally, providing a key protein and micro-nutrient source for over one billion people (World Bank et al., 2012). Despite these drastic numbers, small scale fisheries are often overlooked in terms of resource management and implementation of sustainable practices and policies (Cohen et al., 2019; FAO, 2015; Stevens et al., 2014). This marginalization has resulted in excessive pressures on fisheries (e.g., overfishing, habitat degradation) and vulnerabilities in small scale fishers' livelihoods (e.g. access inequalities, unfair market structures) (Cohen et al., 2019; Jentoft & Midré, 2011; Pauly et al., 2002; Schuhbauer & Sumaila, 2016).

Given that fisheries incorporate complex relations between the environment, society, and economy, finding solutions that ensure the sustainability of marine resources requires understanding and facilitation of social-ecological processes (McClanahan et al., 2009; Silva et al., 2019). Fisheries science in the form of measuring population dynamics and fishing efforts alone will not be sufficient, rather this knowledge must be paired with social science in the form of understanding economic structures and human action, for ultimately fisheries management is about managing people (Bailey et al., 2017; Clay & McGoodwin, 1995). There is no one-size fits-all solution to managing common pool resources, as different ways of governance will be fit for different economic and environmental characteristics (Ostrom et al., 2012). With wide variance among resource pools and resource users, rules imposed from the outside on how to manage social-ecological systems are often unsuccessful (Ban et al., 2009; Bennett & Dearden, 2014; Ford & Stewart, 2021). Rather, community designed problem solving can be more effective, as such solutions are uniquely crafted for the context at hand (Jenkins, 2010; Ostrom et al., 2012). Studies have shown that adoption of sustainable practices are culturally specific and that a focus on individual and community experiences is key to identifying drivers of change, areas of vulnerabilities, and feasible solutions within human-coastal relationships (Allison et al., 2020; Jenkins, 2010; McClanahan et al., 2009; Ostrom et al., 2012). This further emphasizes the importance of stakeholder engagement and social integration with resource management (Chan et al., 2012; Eriksson et al., 2016; McClanahan et al., 2009).

Bycatch is produced when marine animals are incidentally caught in fishing gear. Bycatch of megafauna, including sea turtles, dolphins, and whales, is a priority issue within the conservation sector, with particular calls to collect and monitor bycatch data in small scale fisheries (Lewison et al., 2004; Lewison et al., 2014). Small scale fisheries have historically been seen as more sustainable than industrial fisheries, yet recent studies have shown that gillnet fisheries located in particularly sensitive areas can have megafauna bycatch rates as high as industrial fleets (Alfaro-Shigueto et al., 2010; Lewison & Crowder, 2007; Ortiz et al., 2016; Peckham et al., 2007; Shester & Micheli, 2011). More studies on small scale fisheries megafauna bycatch are being conducted and methodologies used include either observer programs, beach surveys for carcasses, or interviews and surveys with fishers (Moore et al., 2010; Snape et al., 2013). Despite this recent increase in small scale fisheries studies, there remains a consistent call of attention to collect bycatch data (Komoroske & Lewison, 2015; Lewison et al., 2014).

Megafauna conservation, fisheries management, and coastal livelihoods are interconnected in their actions and impacts through their respective efforts, regulations, and practices (Allison & Ellis, 2001; Komoroske & Lewison, 2015). For example, the incidental catch of a turtle can cause damage to fishing nets, alter the rate of catching fish, and increase time at sea for fishers (as observed in Nicaragua, July 2017) (Wang et al., 2010). As such, there appears to be not only an ecological cost to megafauna bycatch, but also a social and economic cost. There are known management tools that could aid in decreasing megafauna bycatch and increasing sustainability of the fishery, yet before implementation of any solution can be put in place, it is critical that a sound baseline be established (Wang et al., 2010, 2013). Fisheries observer programs are the most accurate way to quantify total catch, bycatch, and fishing effort (Babcock & Pikitch, 2003). By deploying observers on board fishing vessels, they are able to collect data at-sea which can then provide the basis for management decisions.

1.1 Study Scope

On a broad level, this thesis aims to understand the logic of marine management strategies and their influence on small scale fisher behavior, and ecological and economic outcomes. To do so, I focus on a case study of the olive ridley sea turtle population in the Chacocente Rio-Escalante Wildlife Refuge (henceforth, the Chacocente refuge) and gillnet fishers from the coastal community of El Astillero in Nicaragua. I develop empirical understanding of fisher behavior and sea turtle bycatch to analyze the effectiveness of the marine management strategies and explore ways to better meet the needs of olive ridley populations and fishers.

There are a limited number of small scale fisheries bycatch studies, and I did not encounter any conducted at the olive ridley nesting beaches of Nicaragua (Lewison et al., 2014). There is little research attention on fisheries in Nicaragua, and what attention there is, is primarily on sea turtle nesting sites and egg poaching in the Pacific and on sea turtles, lobsters, and gleaning activities in the Caribbean (Alvarado & Taylor, 2014; Daw, 2008; González, 2018; Salas et al., 2007). Filling the research gap regarding fishing practices in Nicaragua can then inform which management and capture strategies are best suited to benefit both marine megafauna and fishers, and ultimately sustain this region ecologically and economically (Ortiz et al., 2016). The current priority in the Chacocente refuge is on protecting sea turtle nests and eradicating turtle egg poaching (Hope, 2002). The effort to protect reproductively mature olive ridley adults is via establishment of a marine protected area (MPA) (SINAP, 2008). This study estimates the impact bycatch has on the turtle population, with the hope that resource managers and conservation organizations will be able to weigh this against the impact made by poaching and allocate efforts accordingly (Hamann et al., 2010; Pascoe et al., 2014).

Findings in the fields of environmental social science, social-ecological systems, and fisheries governance, show that resource management from a purely ecological angle is not effective and that inclusion of socioeconomic issues are key factors for success (Bennett et al., 2019; Salas et al., 2007). However, the socioeconomic components of El Astillero and the voices of fishers have not been included in resource management and decision-making processes, as Nicaragua continues to govern from a top-down approach (Crawford et al., 2010; LaVanchy et al., 2020; Solis Rivera, 2012). This study aims to draw attention to the need for incorporating the interests and perceptions of fishers from El Astillero into regional governance strategies.

This thesis consists of four chapters. To follow, is the second chapter, which describes the political ecology and historical context of the Chacocente Rio-Escalante Wildlife Refuge and Nicaragua. Understanding cultural practices, ideologies, and relationships between communities, natural resources, and governing bodies aids in explaining the existence of research and management gaps and leads into the third chapter. This is the main analytical chapter, which examines the logic regarding fisheries regulations within the marine portion of the Chacocente refuge. Through a mixed methods approach, I identify underlying, unintentional management assumptions of fisher-turtle interactions, quantify olive ridley bycatch, and develop understanding of fisher behavior through an at-sea observer program. In looking to the past in chapter two and the present in chapter three, chapter four then looks to the future in consideration of possible approaches to designing management strategies that are ecologically based and human centered.

1.2 Methodology Justification

Strategies for examining the olive ridley turtle population were informed by the International Union for Conservation of Nature (IUCN) Marine Turtle Specialist Group, which calls for decreasing mortality rates through bycatch management, facilitating "training programs for community members… and promoting conservation as an integral part of community development" (Marine Turtle Specialist Group, 1995). These strategies centered our focus to target bycatch through tools which incorporate community participation. Sample and data collection methodologies were adopted from the National Oceanic and Atmospheric Administration (NOAA) and Flora and Fauna International (FFI) observer programs (J. Van Niekerk, J. Bieraugel, A. Cotto, V. Gadea, personal communication, March 2019).

Consideration of the ethical, social, and economic elements associated with working in coastal communities is recognized per the Food and Agriculture Organization (FAO) Voluntary Guidelines for Securing Sustainable Small Scale Fisheries (2015). Guideline No. 2, Respect of Cultures, was acknowledged by incorporating local concerns into the study design and recognizing and working in parallel with existing fishing practices (FAO, 2015). Guideline No. 6, Consultation and Participation, was incorporated by: (1) seeking community input and perspective through open forum meetings, and (2) partnering with community members to carry out the observer program, as it could not be done without the participation of fishers themselves (FAO, 2015). Guideline No. 10, Economic, Social, and Environmental Sustainability, was addressed through the project's core objective to quantify turtle bycatch, in relation to fisher behavior and catch values. Acknowledging the complex social, economic, and political elements that accompany environmental challenges is essential to achieving a triple bottom line in sustainability (social, economic, ecological) (FAO, 2015). Although we do not target each of these elements in the research design directly, we do emphasize economic and social considerations in the contextualization of the case study and discussion of recommendations. Guideline No. 13, Feasibility and Social and Economic Viability, was considered, as assessing the current regulations of, and behaviors within, the fishery can then point to more cost effective and socially appropriate ways to support sea turtle populations and livelihood security (FAO, 2015).

Chapter 2: A Brief History of the Chacocente Rio Escalante Wildlife Refuge

The Chacocente Rio Escalante Wildlife Refuge consists of a marine social-ecological system, located along the Pacific Coast of Nicaragua. To contextualize relationships within the system and to orient towards feasible future directions in management, it is critical to look back on the history and political ecology of this nation and refuge.

Chacocente beach, located within the Chacocente refuge, is one of ten mass nesting sites globally for the olive ridley sea turtle (*Lepidochelys olivacea*) (Marine Turtle Specialist Group, 2008; Plotkin, 2007). Hundreds to hundreds of thousands of females nest in synchronicity at these sites over a one to five day period (Pritchard, 2007). Olive ridley eggs have historically been used as a food supplement and believed by locals to be an aphrodisiac (Campbell, 2007; Hope, 2002). As such, olive ridley eggs carry sociocultural, economic, and ecological importance and the unique mass nesting phenomenon has garnered attention by communities and governments (Campbell, 2007; Hope, 2002; Pritchard, 2007).

The Somoza dynasty and dictatorship took rule of Nicaragua in 1936, and imposed central control over the country's natural resources, driving many of them into degradation (Kinzer, 2007). At this time, Chacocente beach and the turtle eggs laid here were open access – a rarity when all else seemed to be owned in elite hands (Campbell, 2007; Kinzer, 2007). Subsequently, eggs were a staple source of income and food for the seventeen small communities surrounding the nesting site. Hope (2002) states that the, "profits from two nests (of eggs) exceeded what could be earned in a week," and that families decided whether to sell or consume eggs based on market prices and availability of alternative food sources.

In 1979, the leftist Sandinista National Liberation Front (FSLN) overthrew the Somoza regime and in contrast set ecological and social development as central importance to the revolutionary government (Campbell, 2007; Kinzer, 2007). Faber deems the FSLN movement, "ecological socialism" and "revolutionary ecology," and Campbell calls the case of Chacocente in particular one of, "symbolic value" (Campbell, 2007; Faber, 1993). The FSLN planned thirty-six protected areas, yet Chacocente was one of few actually implemented, largely because civil war soon broke out and to-be-protected areas became battle grounds for guerilla warfare (Faber, 2002). The counter group, the Contra, was backed by the United States, while the FSLN was backed by the Soviet Union – making the civil war a proxy war with deeply entrenched principles (Kinzer, 2007). The remote location of, and difficult access to, Chacocente kept it out of use for battle and open for governance. The aim for the Chacocente refuge was, "productive conservation" (Campbell, 2007).

With the release of central power over resources, a land grab ensued. Chacocente beach was squatted by 3,000 people wanting to stake a claim to egg resources, yet the FSLN government issued access rights instead to the families of the original seventeen communities (Faber, 1993). Egg traders, who undercut wages to egg collectors and created a monopsony, were removed as intermediaries and the newly established Ministry for Environment and Natural Resources (MARENA) stepped in to ensure equitable prices (Campbell, 2007; Hope, 2002). Sea turtle conservation became a prominent environmental education campaign nationwide – and always incorporated a component of human use (Campbell, 2007).

As war continued to wage, funds for Chacocente were cut, the Ministry of Natural Resources and the Environment was shrunk, and management dwindled, enabling previously-pushed out traders to step back in and, in ideological accordance, practice their right to livelihood as middle men (Campbell, 2007; Faber, 2002). In 1990, through a peace process, the revolutionary government ended and an anti-FSLN party was elected (Kinzer, 2007). The switch in power brought conflicts over egg resources between squatters, locals, park guards, and the army (Campbell, 2007). In 1993, Chacocente management placed a rotating staff of army and MARENA personnel to be on-site and reestablish protection (this on-site presence remains today). The refuge continued to aim for socially equitable and ecologically sustainable distribution of eggs through a number of strategies including: dividing the beach into "production" and "reproduction" zones; setting season bans on collecting, seasons for consumption-only, and seasons for commercial selling; establishing community and family quotas; and unlimited collections for the first night of an arribada (Campbell, 2007). Ultimately, the dichotomy of some eggs being legal and others being illegal, depending on who was collecting and the time of year of collection, proved difficult to monitor and regulate.

In 2005, Nicaragua banned the collection and trade of turtle eggs and all other turtle products nationally (SINAP, 2008). Conflict between communities with traditional use of this resource and the authorities trying to instill new norms erupted (Campbell, 2007; Faber, 2002). An illegal market for turtle eggs was established, benefitting traders with continued high demand and now low supply, and ladening collectors with risk via illegal sourcing and subsequent chance of fines, arrest, and physical force from authorities (Campbell, 2007; Cornelius et al., 2007).

This management history, rooted in ideology and conflict, has left an indelible legacy (Faber, 2002; Hammack, 2010). In conversations with community members from the fishing town of El Astillero, located a mere five kilometers south of Chacocente beach, many stated they had never visited the refuge, deeming it, "dangerous," and the visitor fee too expensive. They shared stories of relatives being shot at while collecting eggs and alluded to a current resident who was paralyzed during conflict with authorities over eggs. As of 2019, there were no present-day accounts of gunfire. However, there were accounts of physical beatings. In one incident, July 2019, *hueveros,* meaning egg collectors, entered the refuge during the night of a mass nesting event and were caught and beaten by refuge officers. The next day, a refuge officer entered El Astillero and was attacked by one of the *huevero's* brothers in retaliation (field observations, 2019).

Throughout the history of the Chacocente beach and established refuge, a narrative of exclusion from resources, yet belief in access rights to them, permeates. Notions of access rights are tied to people's political ideologies (Faber, 2002). With a deep past of authoritarianism and revolution, and present resurgence of civil conflict as of 2018 with protests and deadly repressions, views of government, resource access, and "ecological socialism," that developed in the 1970s remain pervasive (Faber, 2002; Feinberg, 2018). With the entrenched poverty of this area, access also holds economic value (Hope, 2002). When access has not been given, it has been asserted by some (e.g., as seen through the current illegal presence of *hueveros,* traders, and existence of an illegal market), and then resisted by top-down governance structures (Campbell, 2007; Hope, 2002).

When the Chacocente refuge was established in 1983, the neighboring communities engaged in few ocean-based economic activities (Faber, 1993; Hope, 2002). Instead, communities largely focused on agriculture and livestock activities, and supplemental egg harvests (SINAP, 2008). This historical use of resources, along with conservation objectives to protect olive ridleys, guided the management agenda at the time and resulted in an intensive focus on olive ridley eggs, with very little regard for adult olive ridleys at sea (Hope, 2002; SINAP, 2008).

With political changes in the 1980s and 1990s came reforms in land ownership and use, development of roads and tourism, migration and population growth within the region, and development in fishing (Campbell, 2007; División de Planificación, 2019; Kinzer, 2007; LaVanchy et al., 2020). As of 2019, in El Astillero 60% of households were financially dependent on the fishing industry, with 20% of all employed individuals working within the fishing sector (Luise, 2019). It is arguable that even more households are indirectly dependent on the industry. This is illustrated through conversations with an owner of a local bakery. When asked how their sales were faring, the owner responded, "It was a good morning at sea, so it is a good day here." The owner is alluding to the multiplier effects of fishing, as income generated in the fishery add value throughout the economy (Dyck & Sumaila, 2010; Sumaila et al., 2019). Furthermore, fishing serves as dependable employment for those involved in livelihood diversification (LaVanchy et al., 2020). In 2018, political unrest erupted in Nicaragua (Feinberg, 2018). Travel advisories ensued, along with a drop in the tourism industry (Goett, 2019; LaVanchy et al., 2020). In 2019, one fisher who participated in our study shared how he was previously employed as a cook in a surf destination, called Popoyo, but due to a lack of restaurant customers, he joined a fishing boat in El Astillero to generate livelihood as a crew member. Today, El Astillero is driven by sea based economic activities, with the wellbeing of the community tied to the wellbeing of the fishery.

Fishing activities result in both intentional (i.e., target catch) and unintentional (i.e., bycatch) capture of marine species. Within El Astillero, the catch of adult olive ridley sea turtles is unintentional, yet according to conversations with fishers and review of literature and reports, it does occur, and at a largely unknown rate in this region (Hope, 2002; Lewison et al., 2014; Wallace et al., 2013). Based on species life histories, protecting eggs may not help in sea turtle population recovery if the number of reproductive adults continues to decline (Gilman et al., 2009; Koch et al., 2006; Lewison & Crowder, 2007; Wallace et al., 2010). There is a gap regarding the scale at which bycatch and mortality of adult olive ridleys occurs in and around the Chacocente refuge. Given concern for the health of olive ridley populations globally, and in Nicaragua specifically, filling this gap is critical so that effective management strategies can be prioritized (Eckert et al., 1999; Lewison et al., 2014; Marine Turtle Specialist Group, 1995; Monjeau, 2010; SINAP, 2008).

Managing resources includes the management of human use and behavior (Bailey et al., 2017; Clay & McGoodwin, 1995). Despite changes in use around the Chacocente refuge (e.g., a ban on olive ridley egg harvests, increase in fishing, and subsequent increase in risk to adult olive ridleys via bycatch), management priorities remain the same since establishment, with a focus on olive ridley eggs and an overlook of at-sea activities (Faber, 2002; SINAP, 2008). The identified gap regarding adult olive ridley conservation and fisheries management, and the intensive focus on olive ridley egg management, can be explained by the historical use and political ecology of the Chacocente area. However, for the refuge to meet its original aim of "productive conservation," and to support both human and non-human inhabitants, the gap on bycatch needs to be filled and the effectiveness of management strategies evaluated.

To take a step in closing this gap, we examine the design logic embedded in the Chacocente marine protected area management strategies and ask if these promote the desired fisher behavior and outcomes as concerns olive ridleys. We identify unintentional assumptions regarding fisher-turtle interactions, and challenge them with empirical data via at-sea observations and document analysis. The research design, results, and implications are detailed in the following chapter.

Chapter 3: Intended effects and divergent realities: The Chacocente Marine Protected Area, fisher behavior, and olive ridley bycatch

3.1 Introduction

All seven species of sea turtles are listed on the International Union for Conservation of Nature (IUCN) red list, ranging from vulnerable to critically endangered, with threats including habitat degradation, collection of eggs for consumption, and incidental take of adults through fisheries bycatch (Hope, 2002; Marine Turtle Specialist Group, 1995, 2008). Bycatch, or incidental catch, has been identified as a major driver of population decline for long-lived marine megafauna, including sea turtles (Hamann et al., 2010; Lewison et al., 2004; Soykan et al., 2008). Amidst this risk to marine life are livelihood benefits to humans, as fisheries provide income for over 260 ± 6 million people (Teh & Sumaila, 2013). Such variance between the gains and losses of fishing poses management challenges to meeting livelihood needs of coastal communities and protection needs of marine species.

The olive ridley (*Lepidochelys olivacea*) is the most abundant of all sea turtles; even still, as of 2005, the species experienced a global population decline of 31 to 35% over roughly two generations (i.e., forty years), with an estimated 841,309 nesting females remaining (Marine Turtle Specialist Group, 2008). Females use mixed reproductive strategies, sometimes nesting in solitary, and at other times engaging in synchronized mass nesting events, called “arribadas” in Spanish, which translates to mean “arrivals” (Bernardo & Plotkin, 2007). While females are known to nest in sixty countries, arribadas occur in only six countries (Mexico, Nicaragua, Costa Rica, Panama, India, and Suriname) and at a total of ten beaches, making these select sites ecologically sensitive

(Marine Turtle Specialist Group, 2008). Despite their rarity, arribada rookeries account for the majority of olive ridley nests (Marine Turtle Specialist Group, 2008).

When sea turtles make contact with fishing gear, they can become entangled, lacerated, and in some cases, they are unable to surface and subsequently drown. Lewison et al. (2014) and Wallace et al. (2010) identified the Eastern Pacific Ocean as a bycatch hotspot, given high fishing effort and high megafauna abundance in this region. Specifically, bycatch is thought to be high near Eastern Pacific arribada rookeries, given the mass aggregation of olive ridleys off the shore of the nesting sites, however data on this remains incomplete (Frazier et al., 2007; Lewison et al., 2014; Plotkin et al., 1997). As such, these regions are of particular concern. Sea turtle bycatch occurs most often in shrimp trawl, longline, and gillnet fisheries, with a mortality rate of 37%, 4% to 27%, and 50% to 74%, respectively (Alfaro-Shigueto et al., 2010, 2018; Báez et al., 2019; Dapp et al., 2013; Lewison et al., 2014; Lewison & Crowder, 2007; Liles et al., 2017; Snape et al., 2013). The high mortality in gillnets makes this gear type of particular concern.

Fisheries research has largely focused on the catch, economic production, and environmental impact of large scale fisheries (Dyck & Sumaila, 2010; Norse et al., 2012; Pauly & Zeller, 2016). Small scale fisheries have garnered more attention over the past decade, yet there remains a lack of data on small scale fisheries in general and a gap regarding small scale fisheries bycatch data in particular (FAO, 2015; Gibson & Sumaila, 2017; Hope, 2002; Lewison et al., 2014). Of the few studies quantifying small scale fisheries bycatch, findings suggested the sector generates as much or more bycatch than large scale fleets – an alarming insight that requires attention for conservation, fisheries, and development (Liles et al., 2017; López-Barrera et al., 2012; Peckham et al., 2007).

While bycatch is problematic for sea turtles, it is also a stressor for fishers, as the catch of non-targeted species lowers fishing efficiency and damages gear (Wang et al., 2013). As of 2013, there were an estimated 260 million people employed within the global marine fisheries sector, and of these, 22 million people were employed directly as small-scale marine fishers (Teh & Sumaila, 2013). It was also estimated that 78 percent of marine fisheries workers live in developing countries. Small scale fisheries are clearly critical for livelihood security – as such, improving fishing efficiency is of great importance for human development. With the massive presence of small scale fishers and potentially high production of bycatch, the relationship between small scale fishing strategies and sea turtle populations requires examination.

Conservation actions to reduce indirect take of sea turtles include the establishment of marine protected areas (MPAs). These areas restrict anthropogenic activities (e.g., fishing) to varying degrees with the aim of providing species protection and refuge and subsequently a positive response in biomass and biodiversity (Agardy, 1997; Charles, 2010). For conservation, restricting fishing gear decreases the risk of incidentally catching species of concern. For fisheries, restricting fishing can also result in the increase in biomass of commercially important species, which can then bring "spillover" into neighboring fishing areas, further bringing economic benefit to coastal communities (Edgar et al., 2014; Halpern & Warner, 2002; Sumaila et al., 2015).

Many MPAs diverge from expected outcomes and fail to meet their objectives due to poor engagement and collaboration with local rights and stake holders, high financial cost and physical labor involved in monitoring, and a lack of understanding of ecological and socioeconomic dimensions (Chuenpagdee et al., 2013; Cinner, 2007; Gill et al., 2019; Hilborn et al., 2004; Jameson et al., 2002; Sumaila et al., 2006). As a result, many MPAs are recorded as protected on paper, but not in practice, deeming them "paper parks" (Kaplan et al., 2014). The United Nations World Database on Protected Areas reports that 7% of the ocean is established as a MPA; however, according to the Marine Conservation Institute, only 2.7% of ocean space is protected and managed in practice (Evans, 2020; Sala et al., 2021; World Databasae of Protected Areas, 2022). Therefore, 61% of all MPAs are lacking proper management, enforcement, and not meeting their objectives. Currently, there is a push for 30% protection of the ocean by 2030. While this initiative is positive in theory, we need to build quality protection in practice – not just increase park quantity on paper (Sala et al., 2021). To do so, the MPAs that are already implemented need to be evaluated, their unique social-ecological cases understood, and management redesigned to move towards effectiveness for sea life and communities alike.

This case study focuses on the Chacocente Rio Escalante Wildlife Refuge and the neighboring fishing community of El Astillero along the Pacific coast of Nicaragua. We examine the effectiveness of the established MPA by testing unintentional assumptions in the management design against fisher behavior and outcomes as concerns olive ridley sea turtles.

3.2 Case Study: Chacocente refuge

Chacocente is one of the ten olive ridley arribada sites globally, receiving an estimate of 14,000 to 40,000 nesting females annually (Marine Turtle Specialist Group, 2008; SINAP, 2008) (Fig. 1). In this region, olive ridley eggs have historically been used as a food supplement, believed to be an aphrodisiac and seen as a delicacy – subsequently, they carry social and economic value (Campbell, 2007; Faber, 1993; Hope, 2002). In 1983, the nesting beach and surrounding area was protected as the Chacocente Rio Escalante Wildlife Refuge. According to Article 1 of Decree No. 1294 from 1983, the Chacocente refuge was established "to protect the nesting beaches of the sea turtles *Lepidochelys olivacea* (olive ridley) and *Dermochelys coriacea* (leather back), as well as the last strongholds of the tropical dry forest of the Pacific due to its socioeconomic, ecological, and scientific importance" (SINAP, 2008).

Fig. 1. Map of case study site. Blue pin marks Chacocente Rio Escalante Wildlife Refuge (Esri, 2022).

The refuge encompasses the whole of the arribada nesting beach (1,545 meters), stretches twelve nautical miles to the edge of the Nicaragua territorial sea, and covers roughly 4,645 hectares of land and sea (SINAP, 2008; World Databasae of Protected Areas, 2022). The boundary lines established in 1983 remain the same today (Fig. 2). Six communities live within the refuge, with an additional eleven communities neighboring the refuge, spanning a total of three municipalities and two departments (Campbell, 2007; SINAP, 2008). The wet season stretches from May to October, with rainfall varying depending on elevation and averaging between 800 and 1200 mm per year (SINAP, 2008). For the purposes of this study, scoped by the concerns of fishers in El Astillero, we focus on the marine ecosystem.

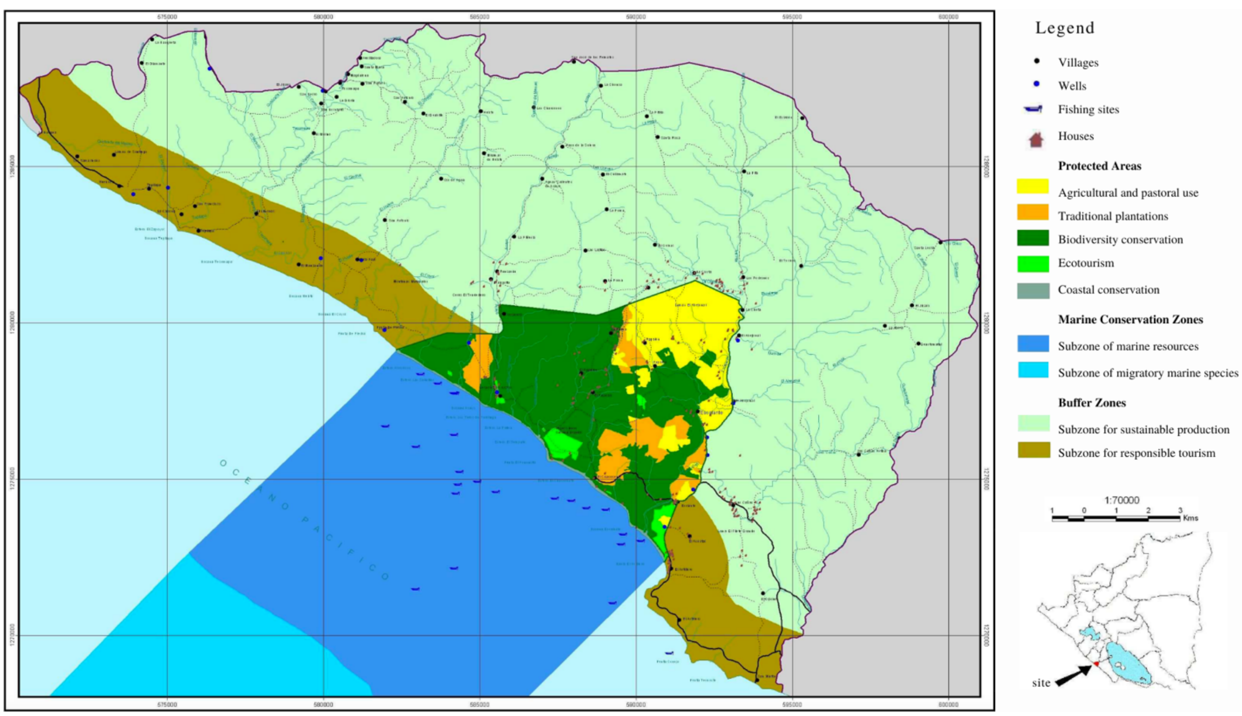

Fig. 2. Zoning of Chacocente Rio Escalante Wildlife Refuge. Adapted from SINAP, 2008.

The marine portion of the Chacocente refuge (henceforth, the MPA) is divided into two zones:

- Subzone of conservation of marine resources;
- Subzone of migratory marine species protection.

The objective of the subzone of conservation of marine resources is to, "regulate small scale and industrial fishing in such a way as to guarantee long-term ecological and economic sustainability of the activity" (Table A.1) (SINAP, 2008). The objective of the subzone of migratory marine species is, "protection of important habitats for the migration of species of international importance and the reproduction and feeding of sea turtles" (SINAP, 2008).

In 2008, The Rio Escalante Chacocente Wildlife Refuge Management Plan was updated, and the main conservation problem identified in the refuge as a whole was the use of natural resources, specifically regarding the use of:

- Eggs of olive ridley sea turtles; and
- Uncontrolled fishing.

Fishing at Chacocente has a shorter history in comparison to egg harvesting. According to Faber (2002), prior to 1979, all but one of the seventeen communities around Chacocente, "had their backs to the sea," and did not engage in ocean-based resources and economies. Over the past fifty years, historic landings of fin fish along the Pacific coast of Nicaragua show an upward trend, with an 85% increase in landed pounds from 1983 to 2019 – the years Chacocente was established and this study was conducted (División de Planificación, 2019; Haas et al., 2015). Fishing practices from the coastal communities of El Astillero, Casares, La Boquita, Huehuete, Tupilapa, Punta de Piedra, and Pie de Gigante, have been identified as a concern given their proximal location to Chacocente and use of gill netting and bomb fishing (SINAP, 2008). The Management Plan recognized that it has not been able to impart influence upon its concern and that the Chacocente MPA has not been properly protected, "due to the costly and difficult regulation at sea" and also that "there is little information on this type of (marine) ecosystem" (SINAP, 2008).

In 2019 in Nicaragua, 36,206 people were employed in the fisheries and aquaculture value chain, and 5,759 were employed in the small scale fishery on the Pacific Coast (División de Planificación, 2019). In the coastal community of El Astillero, which translates to mean, "the shipyard," 60% of households are financially dependent on the fishing industry, with 20%[1] of all employed individuals working within the fishing sector (Luise, 2019). The fishery is small in scale, boats are made of fiberglass, run via an outboard motor, and measure approximately four to five meters in length (Alvarado & Taylor, 2014). Fishers use various fishing gears, including bottom-set gillnets, bottom longline, surface drift longline, hand lines, and spears (INPESCA, 2008). The primary target species include snapper (*Lutjanus* spp.), snook (*Centropomus* spp.), green spiny lobster (*Panulirus gracilis),* dorado (*Coryphaena hypurus*), mackerel (*Scomberomorus* spp.), and corvina (*Cynosciun* spp.) (División de Planificación, 2019; Navarro, 2010; SINAP, 2008).

The Chacocente olive ridley rookery is a mere five kilometers to the north of El Astillero, meaning nearshore fishing efforts are largely in the path of breeding and nesting turtles (Plotkin, 2010; SINAP, 2008). While visiting El Astillero in 2017, fishers verbally expressed to the authors that although they frequently encountered olive ridley sea turtles, they did not want to catch them. They saw bycatch as harmful to the individual turtle, as well as a hassle for themselves requiring more net handling time at sea, net repair time on shore, and unwanted stress. It is illegal to be caught

[1] The number of people who work informally in indirect income generating fishing activities (i.e., baiting hooks, repairing nets, processing catch) is not accounted here (Kasseeah & Tandrayen-Ragoobur, 2015; Teh & Sumaila, 2013). Further study is needed to understand how many people participate in fisheries related activities, their age and gender, and the roles they partake in. This is particularly important to recognize the contributions of women and gender equity in fisheries (Harper et al., 2013; Thomas et al., 2021).

with a turtle, with penalties including arrest, fines, or physical force once back on shore (SINAP, 2008).

As mentioned, the MPA's objective is to regulate artisanal and industrial fishing and reduce threats to migratory species (SINAP, 2008). To meet these objectives, certain marine activities are permitted, and not permitted within the MPA boundaries. Permitted activities include small scale fishing with vertical bottom longline, bottom hand line with single or multiple hooks, and fishing with rod and hook. This is only permitted for communities surrounding the refuge. Catch and release sport fishing and recreational diving are also allowed within the MPA. Prohibited activities include small scale fishing with lobster pots, fish pots, surface gillnets, midwater gillnets, bottom gillnets, beach seines, and any equipment that drags on the surface, midwater, or bottom. Extraction of sea turtles and their eggs, sharks, whales, dolphins, rays, and any species on the IUCN Red List or Convention on International Trade in Endangered Species of Wild Fauna and Flora (CITES) appendices is prohibited at any scale. Extraction of lobster, shrimp, and crab is prohibited at an industrial scale. These regulations exist year round (SINAP, 2008). In summary, small scale fishing with pots, gillnets, trolling, and trawling, and the capture of internationally protected species, is not allowed at any time within the MPA boundary.

3.2.1 Research Questions

The existence of a 36-year-old MPA with the objective of protecting olive ridleys, along with the present expressed concern for uncontrolled fishing and turtle bycatch by managers and fishers, respectively, inspired the broad research questions:

- Do the design logics embedded in the MPA management strategies, specifically regarding small scale fisheries, promote the desired behavior and outcomes as concerns protection of olive ridleys? If so, how or why not?
- Can the Chacocente MPA be better tailored to meet the needs of both olive ridley sea turtle populations and fishers? If so, how?

Fishers of El Astillero stated that sea turtle bycatch occurred most frequently in the gillnet fishery, which is also consistent with the high mortality rates for turtles caught in gillnets reported in literature (Lewison et al., 2014; Liles et al., 2017; López-Barrera et al., 2012; Shester & Micheli, 2011). Yet, sea turtle bycatch and mortality in gillnet fisheries in Nicaragua has yet to be quantified, and specifically remains undocumented along the Pacific coast in and around the Chacocente refuge. The scale of this problem is unknown, and as such, the impact of bycatch on olive ridley population health is unknown. Understanding the scale and impact of present threats to the olive ridley is pivotal knowledge to guide effective management and conservation strategies (Alfaro-Shigueto et al., 2018; Hamann et al., 2010; Wallace et al., 2013).

Given the aforementioned research questions, concerns for gillnets, and gap on bycatch, the focus of this study is the Chacocente MPA's prohibition regarding the use of gillnets. Gillnets are used to fish for a variety of species, with subsequent variation in gear design and fishing strategies, yet MPA regulations make no distinction regarding different targeted species (INPESCA, 2008). The only species-specific regulation is that fishing for animals listed on IUCN and CITES is not permitted. Furthermore, MPA regulations are in place all year round, yet fishers understood regulations to only apply during the turtle nesting months, from June to November.

Regulations within the MPA stem from the objective to reduce threats to migratory species, particularly olive ridleys (SINAP, 2008). As such, the prohibition on gillnets seems to presume the idea that: limiting gillnetting spatially (nearest the nesting beach) will reduce turtle bycatch; prohibiting gillnetting for all targeted species is necessary to reduce turtle bycatch; prohibiting gillnetting in all nesting months is necessary to reduce turtle bycatch. Together, these briefly summarize the apparent unintentional assumptions regarding the interactions and threats between fishers and olive ridleys, which include spatial, gear, species, and temporal factors.

To answer our higher-order research questions, we evaluate the unintentional management assumptions against empirical data with four more specific research questions:

1. Has the implementation of the MPA reduced gillnetting nearest the nesting beach and so too the associated bycatch?
2. Has prohibiting gillnets encouraged the use of alternative, permitted gear?
3. Is gillnetting, for any species threatening to olive ridleys?
4. Is gillnetting, during any nesting month threatening to olive ridleys?

The regulations and derived questions informed the research design as follows in Table 1 and as described in the methods section.

Table 1. MPA regulations on small scale fishing, derived underlying and unintentional assumptions, and subsequent research questions. Unintentional assumptions were tested against observations from at-sea observer program in 2019 (SINAP, 2008).

MPA Regulations	**Regulations**	**Unintentional Assumptions**	**Research Questions**	**Observations Sought**
Not Permitted – Small scale fishing under the modalities of setting pots for fish and lobsters, *laying of surface, drift, mid-water or bottom gillnets,* beach seines and any fishing method that involves dragging fishing equipment on the bottom, to midwater or shallow	Spatial – no gillnets are permitted within the MPA	If the MPA is established, then gillnetting near the nesting beach will decline, and so too olive ridley bycatch	Has the implementation of the MPA reduced gillnetting nearest the nesting beach and so too the associated bycatch?	Fishing activity and olive ridley bycatch events inside and outside the MPA
	Gear type – gillnets are not permitted, yet vertical longline, hand line, and rod and hook are permitted	Gillnets are a threat to olive ridleys – if gillnets are banned, then alternative gear will be used	Has prohibiting gillnets encouraged the use of alternative, permitted gear?	Gear types used by fishers
Permitted – Small scale fishing under the vertical bottom longline modality, bottom hand line with single or multiple hook and fishing rod with hook, valid for the communities surrounding the refuge	Target species – gillnetting for fish and lobster, of any species, is not permitted	Gillnetting for any species is threatening to olive ridleys	Is gillnetting for different species threatening to olive ridleys?	Olive ridley bycatch events while gillnetting for snapper, snook, lobster
	Temporal – regulations apply all year on paper, yet are understood to apply during the turtle nesting season (June to November) by fishers	Gillnetting during any arribada nesting month is threatening to olive ridleys	Is gillnetting, during any arribada nesting month, threatening to olive ridleys?	Olive ridley bycatch events while gillnetting from June to November

3.3 Methods

This research used the Chacocente MPA and the El Astillero gillnet fishery as a case study to evaluate the effectiveness of management practices. A case study approach, consisting of empirical data on a phenomenon, enables the capture of unique complexities within a bounded scope (Elson et al., 2018; Hyett et al., 2014; Johansson, 2007). Given the distinctiveness of coastal communities and small scale fisheries (i.e., political histories, cultural practices, and ecological conditions), and the rarity of olive ridley arribada sites, a case study approach was appropriate to address our research questions (Campbell, 2007; Teh et al., 2015). This study focuses specifically on fisher-turtle interactions and the ecological and socioeconomic context in which they occur along the Pacific coast of Nicaragua. Data was collected through a mixed methods approach, with novel empirical data via at-sea observations on fish catch, bycatch, and catch value, participant observations in the field, and qualitative document analysis. The stages of work and their timelines are depicted below in Figure 3.

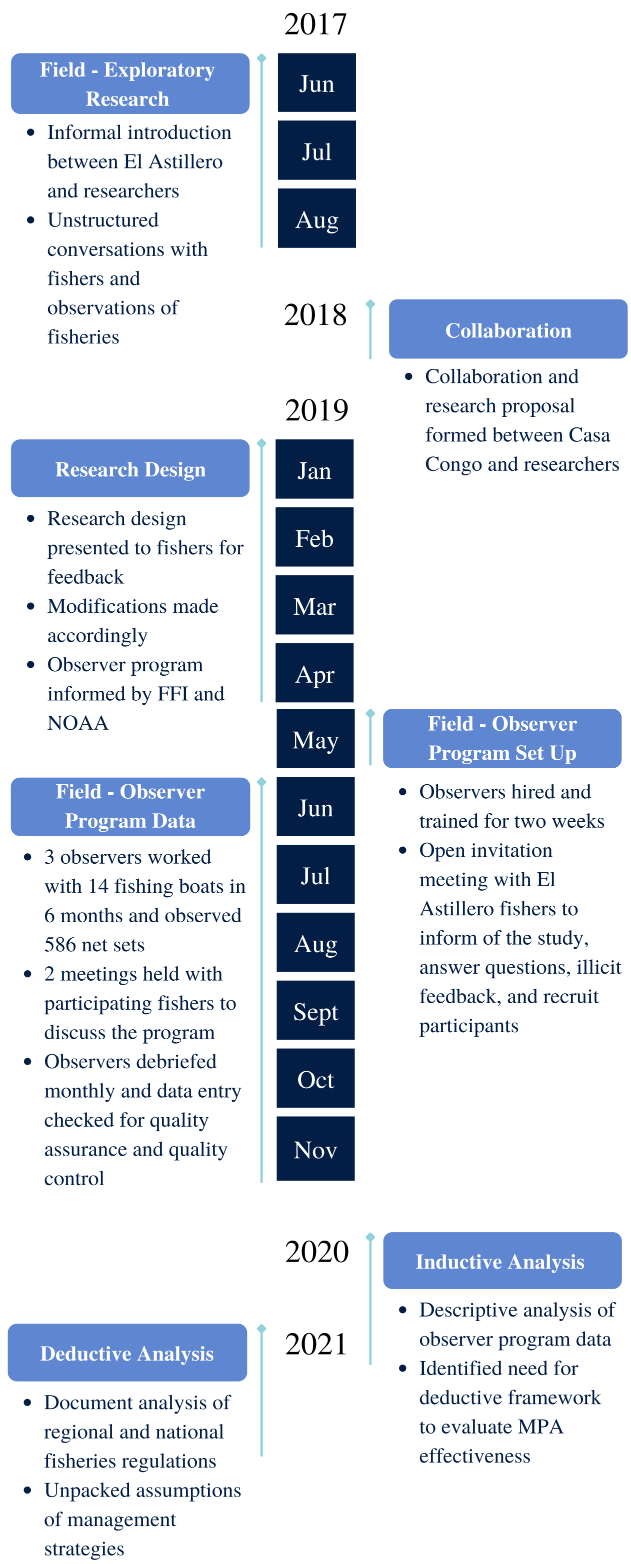

Fig. 3. Study timeline and stages.

3.3.1 Exploratory Research

Exploratory research was conducted in 2017 through informal observations of, and conversations with, managers, fishers, and community members. This informed understanding of community concerns and interests, with common themes including olive ridley sea turtle bycatch, decreasing fish catches, limited market access, and ineffective marine management. These conversations led to a collaboration between researchers, fishers, and a local non-governmental organization involved in sustainable development, called Casa Congo.

3.3.2 Observer Program

Megafauna bycatch data in small scale fisheries is poor overall, and absent in this case (Hope, 2002; Lewison et al., 2014). To answer our research questions, we aimed to quantify bycatch and begin construction of a baseline. Observer programs (i.e., as opposed to those based on self-reported behavior) are known to provide the highest quality bycatch data, but they are also compromised by the sheer difficulty of perfect versus willing enrollment of fishers (Babcock & Pikitch, 2003; Lewison et al., 2004). With this in mind, we implemented a program in El Astillero. Observer program design was adapted from the National Oceanic and Atmospheric Association and Flora and Fauna International and an early-stage meeting was held with collaborating fishers for feedback on the design (Brooke, 2015).

An open invitation (i.e., via posters at local shops, fish markets, and sports field, and door-to-door visits to fishers' residence) was extended to El Astillero fishers to join a meeting prior to program implementation. The meeting entailed explaining the purpose of the project, discussing interests and concerns, and recruiting participation. Only gillnet fishing was included in the study, and out of the 30 captains in the gillnet fleet, 14 volunteered to participate (Babcock & Pikitch, 2003). Three high school graduates of El Astillero, who had completed programs in environmental education through Casa Congo, were hired as at-sea observers and trained for two weeks in data collection, fish species identification, and sea turtle handling (Northwest Fisheries Science Center, 2019; Pugliares-Bonner et al., 2007; Razzaque et al., 2019). Captains took observers onboard their fishing vessels to collect catch data and participated in two meetings during the course of the program to check in on the observer program and ask if adjustments needed to be made. Participants were thanked for their time and effort with monetary compensation to be used for fishing gear repair. Observers were debriefed each month to review data collection procedures to control for bias, and to check in on the program and ask if adjustments needed to be made (e.g., methods of coordinating with captains, at-sea work load) (Northwest Fisheries Science Center, 2019).

The program ran from June to November of 2019, covering the span of the olive ridley nesting season. We observed 586 net sets, totaling 138,310 meters in net length, over 98 fishing trips, targeting three different species including snapper, snook, and lobster. Data were gathered on the time, location, and gear specifications of fishing practices, and the species, size, weight, and market value of both intentional fish catch and unintentional sea turtle bycatch (Brooke, 2015; Northwest Fisheries Science Center, 2019; Wang et al., 2010).

With the unpredictable and varied nature of fishing in general, and small scale fisheries in particular, our sample was necessarily convenient sampling. As many observations as possible were collected, but this was always contingent on the captain's approval, which was contingent on proper function of out-board motors, gear being repaired, abundance of fish, fair weather conditions, and fair standing with middle men (i.e., fishers are commonly indebted to middle men, who hold access to ice, gasoline, and buyers – these variables influence risk and the decision to fish) (Babcock & Pikitch, 2003). Fishing activity and the targeting of certain species fluctuated over the sample period. The figure below illustrates the distribution of the sample by month and species (Fig. 4).

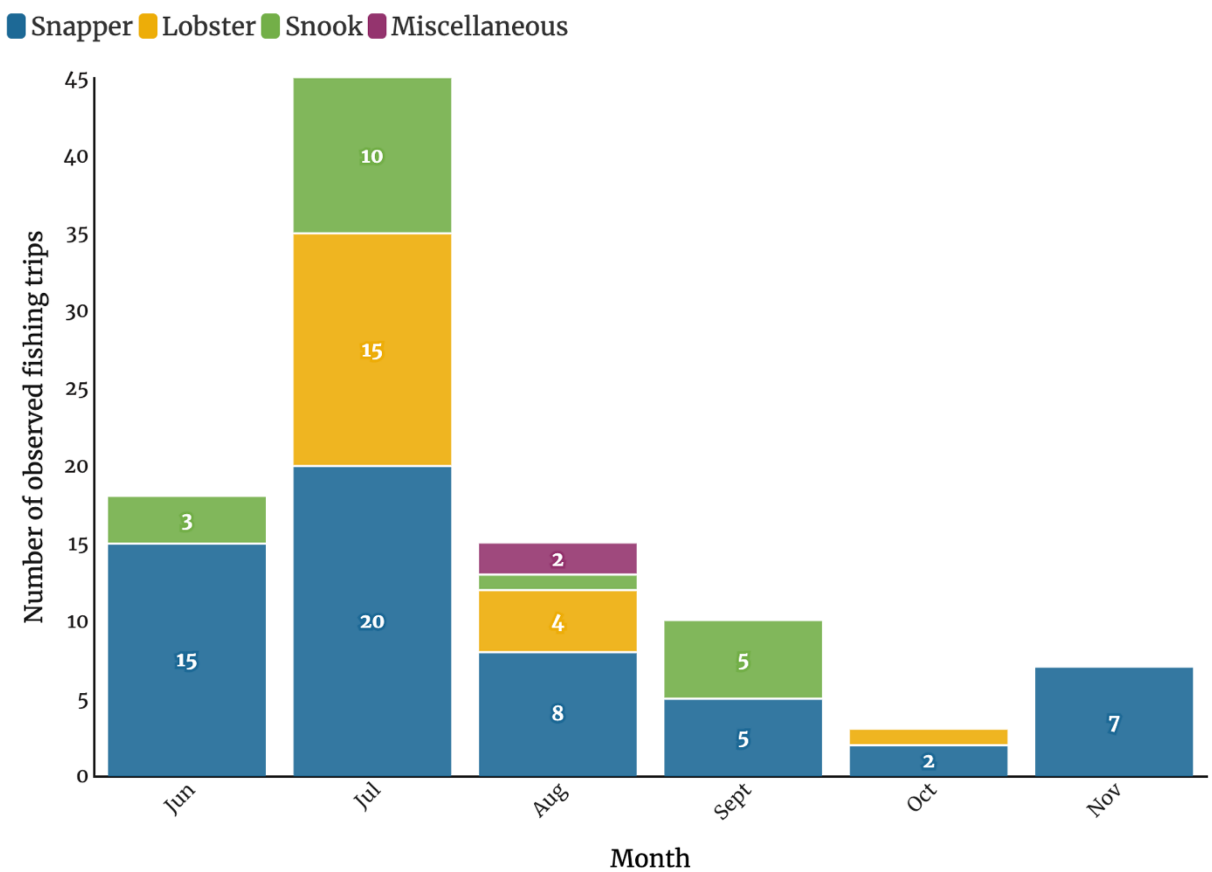

Fig. 4. Distribution of observed fishing trips in 2019 by month and targeted species.

There is no baseline data on the number of olive ridley nesting females and specific fishing activities before the MPA was implemented, and so a before-and-after comparison is not feasible for measuring effectiveness. To begin construction of a baseline, the original research design included a multi-year observer program and additional trip to the field, but these activities were cancelled in the interest of public safety during the COVID-19 pandemic. While our 2019 sample is small, it is representative of 46% of the El Astillero gillnet fleet.

3.3.3 Document Analysis

In addition to the data from the field, this study employs document analysis to understand the particular purpose of, and unintentional assumptions within, MPA management strategies. Using document analysis of the Chacocente Management Plan of 2008, we isolate all strategies involving fishery prohibition (Coffey, 2013; Siegner et al., 2018). Typically, as was also the case here, regulations on small scale fishing restrict behaviors spatially (where fishing can or cannot take place), on gear use (nets and boats used), on targeted species (allowed or disallowed), and temporally (when fishing is or is not allowed) (Agardy, 1997; Edgar et al., 2014). As such, the overall underlying assumption seems to be that if these prohibitions are minded, then threats to olive ridley sea turtles will decrease.

Additional secondary source document analysis was performed on Nicaraguan fisheries policies and reports, and reports on the state of olive ridley sea turtles and fisheries livelihoods from NGOs. Although the document analysis of these additional reports is not featured in the research question development, they were vital to interpreting and triangulating the findings from the assumptions testing.

3.3.4 Data Analysis

To answer the four questions that interrogate the unintentional assumptions in the MPA regulations, we used empirical data from the at-sea observations in 2019 (Table 1). We selected variables and conditions (e.g., location of fishing and bycatch events, gear types used, and changes to bycatch when targeting specific species and at different times) to examine the effectiveness of the MPA using the assumptions.

For example, to see if spatial assumptions in the MPA design affected fisher behavior and subsequently if this provided protection to sea turtles in the ways expected, we examined where fishers fished and where bycatch events occurred in relation to the MPA boundary lines. To see if gear assumptions affected fisher behavior, we examined what percentage of the entire fishing fleet use gillnets. To see if unintentional species assumptions held true, we examined the relationship between the species fishers targeted and bycatch events. To see if unintentional temporal assumptions held true, we examined the relationship between monthly fishing activity and bycatch events.

The statistical and spatial analysis were conducted in Microsoft Excel 16.61 and ArcGIS online. To better understand the variation in our data, we cross checked observer program data and national fish landing data from the Nicaraguan Institute for Fisheries and Agriculture (INPESCA) and found parallel variations (División de Planificación, 2019). The variation in our sample, regarding targeted species and seasons, is explored further in sections 2.4.3 and 2.4.4.

In order to understand the rate, or the effectiveness, at which turtle bycatch, fish catch, and catch value occurs within the gillnet fishery, we held our observed data against a constant unit of fishing effort, that being 100 meters of gillnet (Peckham et al., 2016; Wang et al., 2013). Holding a constant unit enabled us to examine the relationships between bycatch, catch, and value, particularly to see if the occurrence of bycatch affected fish catch and the generation of revenue, and to see how these relationships compared across space, species, and time. We calculated catch, bycatch, and value per unit effort as:

Bycatch per unit effort (BPUE) = # turtles captured / (net length/ 100 meters)

Catch per unit effort (CPUE) = lbs. landed fish / (net length / 100 meters)

Value per unit effort (VPUE) = $USD revenue earned / (net length/ 100 meters)

It must be noted that based on field observations, a varying portion of landed catch was reserved by fishers as food for their families and to gift to community members. This is subsistence catch and it carries nonmonetary value, with contributions to food security and cultural practices (Harper et al., 2020; Pauly & Zeller, 2016; Zeller et al., 2006). We acknowledge the importance of subsistence catch, however, the monetary value of this catch was not accounted for in this study due to difficulty in tracking changes in fish biomass from landings, to cleaning, to gifting, to home, and to market. For reliability and validity, pounds of landed fish (to calculate catch per unit effort) was collected as at-sea retained catch (i.e., both targeted species and incidental species that were caught and retained in the vessel to bring ashore). Revenue earned (to calculate value per unit effort) was collected as ex-vessel value (i.e., retained catch separated by species and grade, then the price per pound at first purchase in the commercial market, multiplied by total pounds sold)

(Sumaila et al., 2015). As such, value per unit effort captures purely monetary value via market sales and revenue generated.

3.4 Results

This section reveals our findings from testing the unintentional assumptions of the MPA regulations against empirical at-sea observations from 2019. The assumptions of the MPA are associated with research questions. Answering these four research questions develops understanding of the interactions between fishers and olive ridley sea turtles. This then enables us to answer our broad research questions, regarding the extent to which design logics in the MPA are upheld by behavior and outcomes, and opportunities for the MPA to be better tailored to meet the needs of turtles and fishers. Answers to these broad research questions are explored within the discussion.

3.4.1 Unintentional Assumption 1

If the MPA is established, then gillnetting near the nesting beach will decline, and so too olive ridley bycatch.

Ultimately, the implementation of the MPA has not eliminated gillnetting nearest the nesting beach and the associated bycatch. While slightly less gillnetting occurred inside the MPA boundary compared to outside, the same number of bycatch events occurred inside and outside the MPA boundary. Of all observed fishing activity, 43% occurred inside the MPA, and 57% occurred outside the MPA (Fig. 5). In total, 50 olive ridley bycatch events were observed, and two of those events did not have spatial data. Spatial data is indicative of fishing and bycatch inside and outside the MPA boundaries, and so without it the activities orientation in relation to the boundaries is inconclusive. As such, fishing and bycatch events without spatial data are eliminated for this portion of analysis. Of the 48 bycatch events with spatial data, 50% occurred inside the MPA boundary, and 50% occurred outside, with 24 turtles caught inside and outside, respectively. In some instances, a single net set caught more than one turtle. Inside the MPA, the 24 turtles were caught in 16 different net sets. Outside the MPA, the 24 turtles were caught in 17 different net sets. Mortality rate of turtle bycatch was 79%.

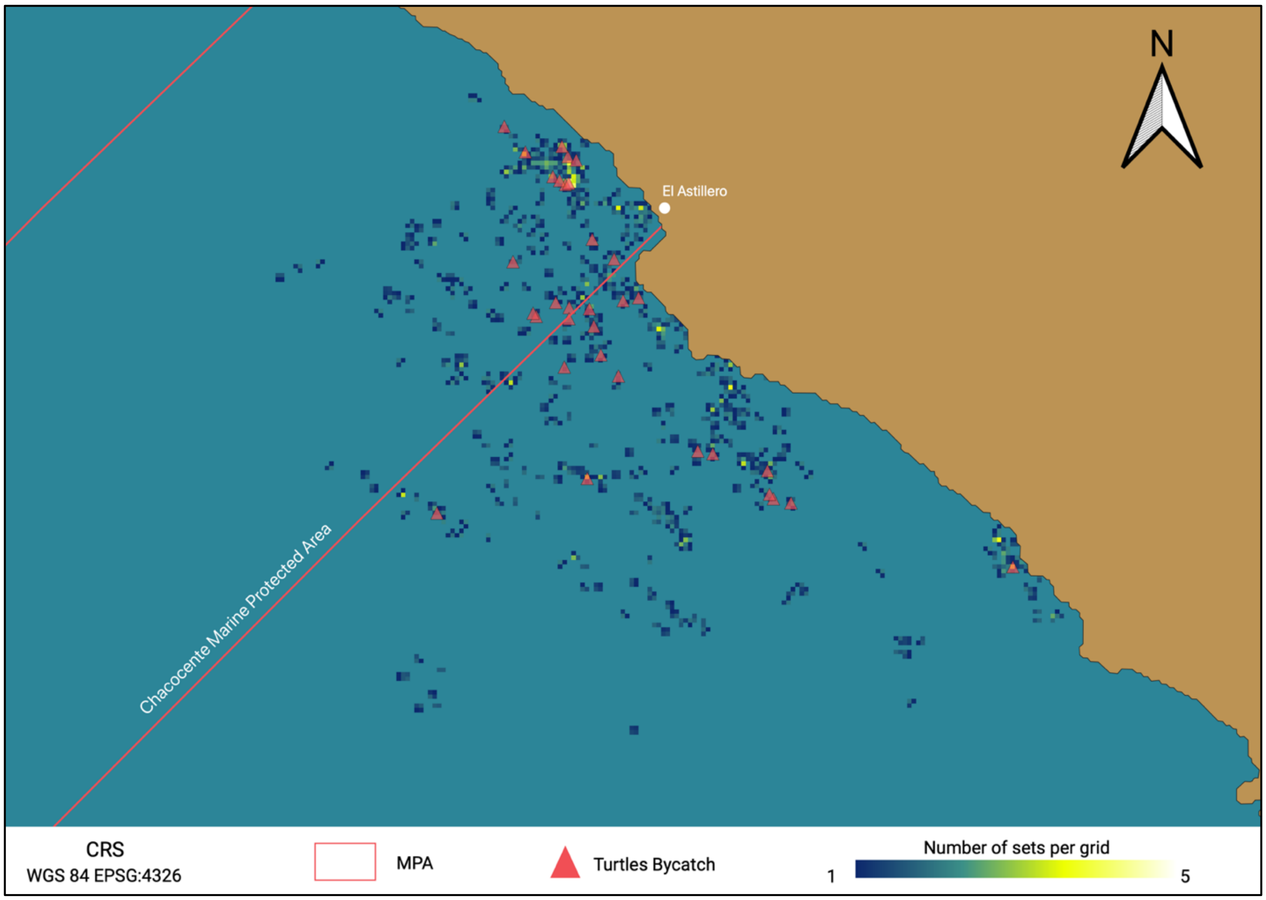

Fig. 5. Spatial density estimation of observed fishing net sets. Dark blue grids represent one net set, white grids represent areas where up to five net sets were deployed. Red triangles represent net sets with one or more turtle bycatch events. MPA coordinates sourced from World Database Protected Areas, 2022; net set data sourced from at-sea observations 2019; map created in Esri, 2022.

Results are reported as proportions to understand behavior and outcomes relative to the whole of observed fishing activity. We also calculated bycatch per unit effort, catch per unit effort, and value per unit effort to understand effectiveness of fishing efforts as stand-alone rates. Bycatch per unit effort is slightly higher inside the MPA than outside, with 0.4 turtles caught per unit of effort inside, and 0.3 turtles caught per unit of effort outside (Table B.1). Conversely, catch per unit effort and value per unit effort are slightly lower inside the MPA than outside (inside: 4.56 lbs. fish caught per unit effort, \$2.36 dollars earned per unit effort; outside: 5.6 lbs. fish caught per unit effort, \$2.79 dollars earned per unit effort). Results indicated that catch per unit effort and value per unit effort have a low positive correlation, $r(91) = +.35$, $p < .001$, and that there is no significant linear correlation between catch per unit effort and bycatch per unit effort, $r(91) = -.07$, $p = .48$. This was as expected: that an increase in catch would be associated with an increase in catch value. The lack of a clear positive relationship between catch and bycatch is in alignment with what fishers expressed regarding the negative effects of bycatch and how entanglement can prevent the net from fishing properly for the targeted species. According to our results, there is no incentive, by way of an increase in catch, to produce sea turtle bycatch.

Having established that fishing and bycatch occur both inside and outside the MPA, for the remainder of analysis we consider all observations, not merely those within the MPA, in order to analyze a larger sample.

3.4.2 Unintentional Assumption 2

Gillnets are a threat to olive ridleys – if gillnets are banned, then alternative gear will be used.

Gillnets are widely used inside and outside the MPA. Prohibiting gillnets has not encouraged the use of alternative permitted gear. Half of all fishers in El Astillero use gillnets as their primary fishing gear, meaning this is the gear type they fish with the majority of the time. This finding is based on self-reports from fishers. In 2019, there were 60 active fishing boats, 30 of which identified gillnets as their primary gear type. In 2021, there were 82 active fishing boats, 40 of which identified gillnets as their primary gear type (Casa Congo, 2021).

As the overall fleet has increased, the proportion of gillnetters remains relatively constant – despite a ban on gillnetting within the Chacocente MPA. Based on historical records, fishing activity along the Pacific Coast has increased steadily since the MPA establishment (Fig. 6) (Haas et al., 2015). However, we do not have baseline data to know how the use of gillnets specifically in El Astillero has changed since the MPA establishment.

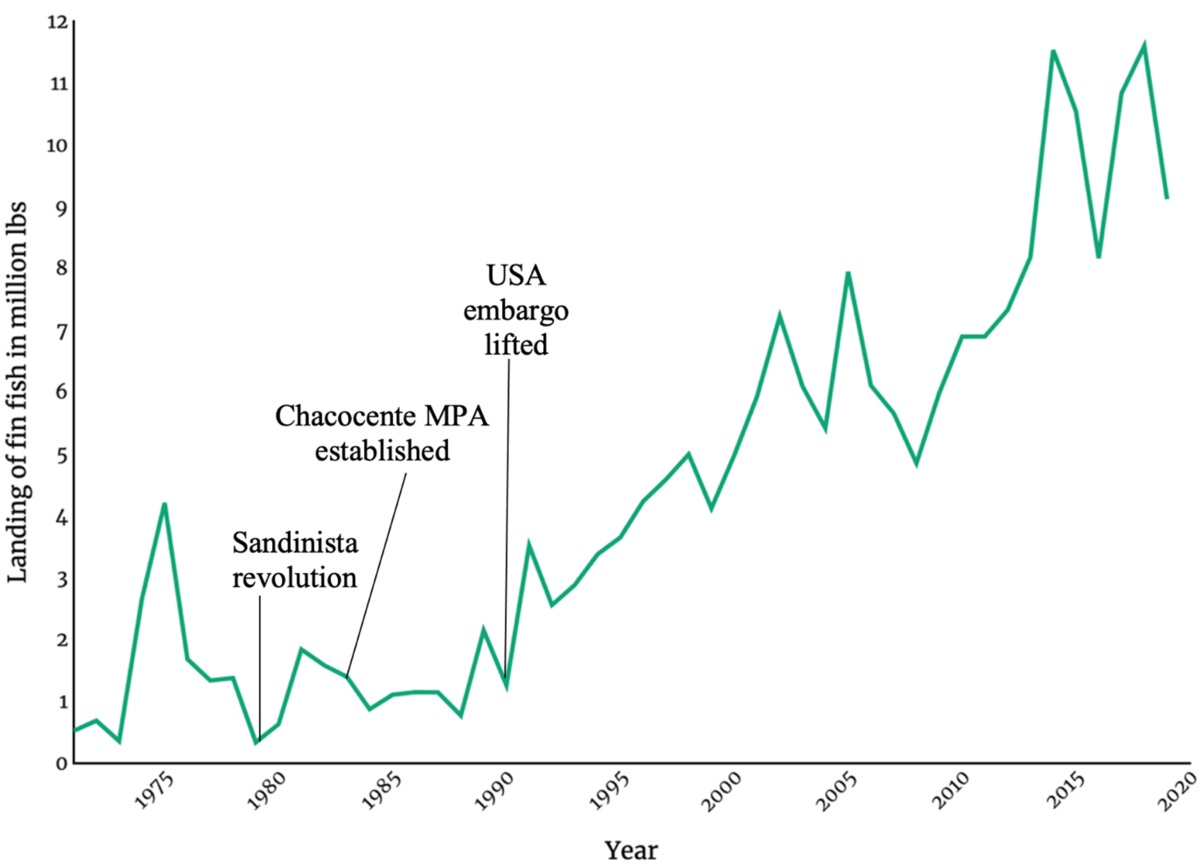

Fig. 6. Historic fin fish landings from the Pacific coast of Nicaragua, with records spanning from 1971 to 2019 (División de Planificación, 2019; Haas et al., 2015).

Gillnetting is conducted in the morning for two to five hours in nearshore waters. Nets are set one morning, left to fish for twenty four hours, and picked up the next morning. Based on field observations, alternative fishing gears used in El Astillero include surface and bottom longline, hand line, and spear. Longline and hand line require the purchase of bait or ownership of an additional gillnet to catch bait fish, twelve to twenty-four hours of time at sea, lights to fish through the night, and are conducted further offshore at approximately ten kilometers and require more gasoline to access. Spear fishing in this area is commonly performed with an air compressor for diving. These alternatives carry additional financial costs and a variety of risks compared to gillnetting. Furthermore, surface and bottom longlining and spear fishing are also not permitted in the Chacocente MPA according to regulations (SINAP, 2008). Permitted alternatives are hand line, vertical longline, and rod and hook. In our observations, hand line fishing is conducted in El Astillero, yet it is used minimally compared to gillnetting and longlining. We did not hear of or encounter the use of vertical longlines, and rod and hook fishing was only encountered as sport fishing activities for tourists.

Nicaragua is the second poorest country in the Western Hemisphere based on GDP, and small scale fishers are amongst the most marginalized in the nation (Cotto & Marttín, 2007; LaVanchy et al., 2020; Solis Rivera, 2012). Financing options available to fishers are limited, and furthermore fishers are subject to debt with middle men due to unfavorable market structures for producers (Alvarado & Taylor, 2014).

3.4.3 Unintentional Assumption 3

Gillnetting for any species is threatening to olive ridleys.

Bycatch threats to olive ridleys vary when targeting different species. At-sea observations show gillnetting for snook produces relatively high bycatch in comparison to snapper and lobster, and between snapper and lobster the difference is negligible. Based on raw data, out of all turtle bycatch events, snapper fishing caught 12% of the turtles, snook fishing caught 55%, and lobster caught 33% (Fig. 7). Not all species are equally sought throughout time, creating changes in fishing effort and subsequently in our observation sample size. To account for this, we normalized turtle bycatch by the observed meters of net set for each target species. This shifts the proportion, with snapper accounting for 5% of turtle bycatch events, snook fishing accounting for 74%, and lobster accounting for 21%.

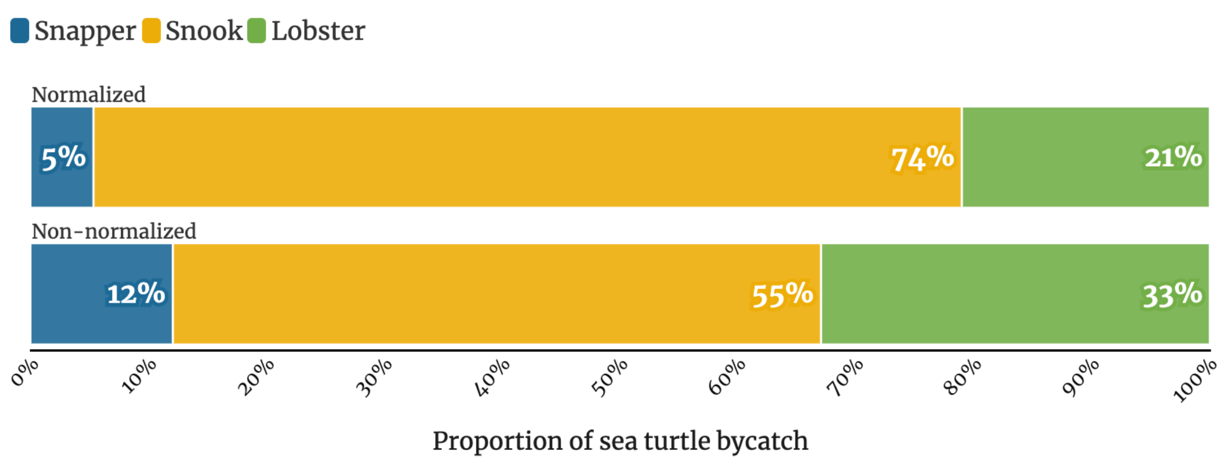

Fig. 7. Proportion of bycatch by targeted species (non-normalize). Proportion of bycatch by meters of net set for each targeted species (normalized).

We calculated bycatch per unit effort in order to understand the rate of incidentally catching a sea turtle while targeting each species. Snook showed the highest bycatch per unit effort, at 0.14 turtles caught per unit effort, followed by lobster at 0.04 turtles caught per unit effort, and then snapper at 0.01 turtles caught per unit effort (Table B.2). These independent bycatch rates align with the relative proportions of bycatch, with gillnetting for snook accounting for the highest amount in both cases. To understand if there is a difference in bycatch rates while targeting each species relative to the MPA boundary lines, we then examined bycatch per unit effort inside verses outside the MPA and found rates were equivalent for snapper (inside: 0.01; outside: 0.01), slightly lower for snook (inside: 0.12; outside: 0.17), and slightly higher for lobster (inside: 0.05; outside: 0.03).

It is important to note the economic value of these target species given the contribution of fishing to community livelihood. Value per unit effort from trips targeting snapper generated \$3.17 per unit effort, snook generated \$3.06, and lobster generated \$1.17 (Table B.2). Based on catch values of all observed fishing trips, sale of snapper generated 40% of revenue, snook generated 15%, and lobster generated 12%. While snapper and snook generate similar value per unit effort, the catch and sale of snapper occurs at a greater proportion and so generates a greater percentage of revenue overall.

This local observation is also found on a national level. According to INPESCA, the total pounds of fin fish landed on the Pacific Coast in 2019 was composed of 39% snapper and 5% snook (lobster was not included in this compositional breakdown, as it is a crustacean) (División de Planificación, 2019). In both our local study observations and the national reports, the proportion of snapper production is constant, while snook is different, by 10%, when comparing local and national findings. Discrepancies could be due to landing data stemming from different points in the value chain (i.e., producer vs exporter) and time (i.e., six month vs annual). Regardless, the catch value from snook remains lower than snapper.

Variation in turtle bycatch proportions and rates can likely be explained by the different strategies used to catch different species, most notably, the size of the mesh that makes up the gillnet (Fig. 8). Gillnets function by forming a vertical mesh wall in the water column, and as fish swim into the wall, the mesh ensnares the mouth, head, gill cover, or midbody. Mesh size and targeted species are moderately positively correlated, $r(558) = .56$, $p < .001$. When targeting snapper and lobster, fishers use gillnets with mesh size 3.5" to 5" and when targeting snook they use mesh size 7".

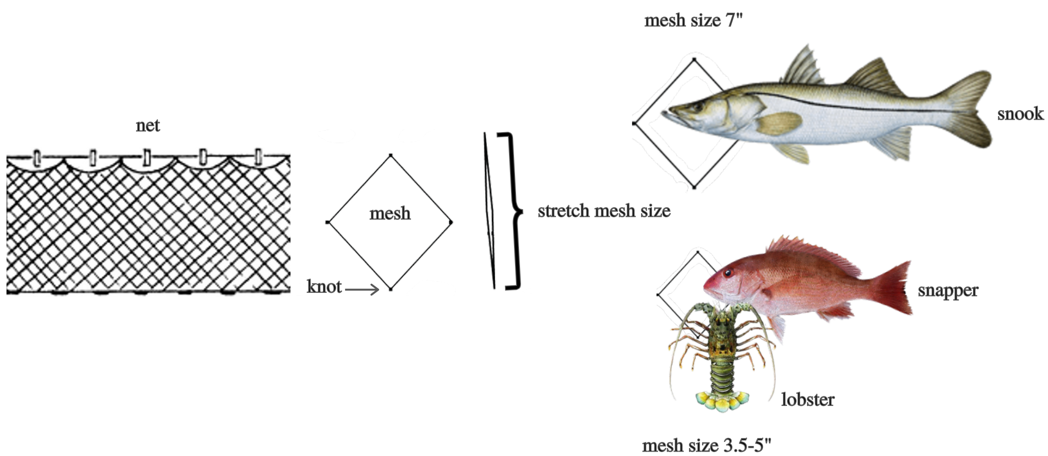

Fig. 8. Gillnets have a float line on top, lead line on bottom, mesh in between. Mesh is measured at maximum diagonal width, called stretch mesh size. Snook targeted with mesh size 7", snapper and lobster targeted with mesh size 3.5-5" (Pugliares-Bonner et al., 2007; INPESCA, 2019).

3.4.4. Unintentional Assumption 4

Gillnetting during any arribada nesting month is threatening to olive ridleys.

While the MPA regulations close gillnet fishing all year round, the fishers understood the closure to be June through November, during the turtle nesting season. While this discrepancy in information is interesting and problematic in and of itself, we examined the nesting months to see how the presumed MPA design affected fisher behavior and outcomes on turtle bycatch.

Bycatch threats to olive ridleys vary across arribada nesting months. Observations show gillnetting in September produces relatively high bycatch in comparison to all other nesting months. Of all observed olive ridley bycatch events, 42% of them occurred in September, 40% occurred in July, and less than 10% occurred in every other month (Fig. 9). To account for the variation in our sample, stemming from variation in fishing activity over time, we normalized bycatch data by the observed meters of net set in each month. This shifts the proportion, with September accounting for 66% of bycatch events, July dropping to account for 10% of bycatch, and October rising to account for 10%.

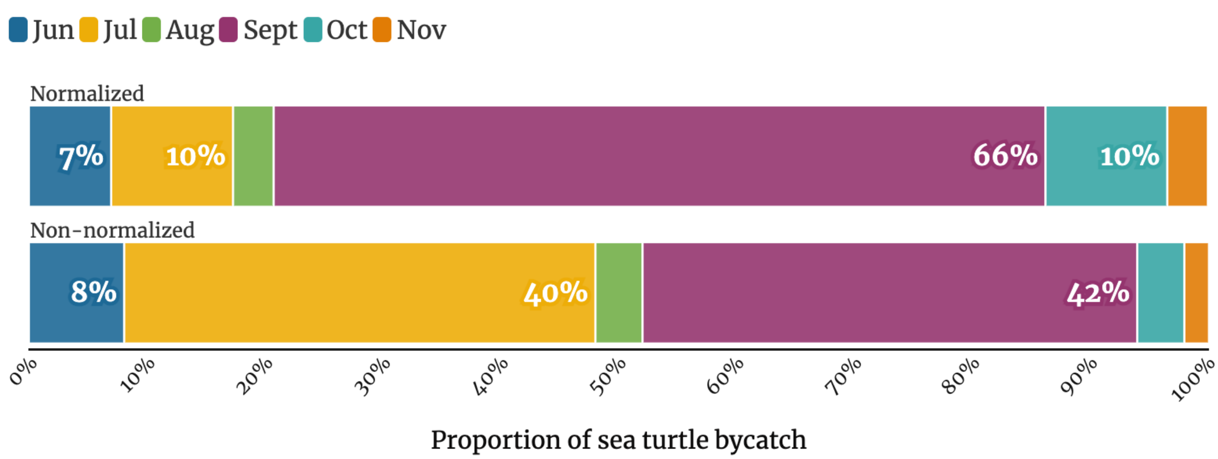

Fig. 9. Proportion of bycatch by month (non-normalized). Proportion of bycatch by meters of net set for each month (normalized).

To understand the rate of incidentally catching a sea turtle while gillnetting in different months, we calculated bycatch per unit effort. Findings were consistent with non-normalized and normalized proportions, with September having the highest bycatch per unit effort at 0.19 turtles caught per unit effort, followed by October and July, both at 0.03 turtles caught per unit effort (Table B.3). We then examined bycatch rates inside and outside the MPA boundary and found September (inside: 0.92; outside: 0.10) and October (inside: 0.19; outside: 0.00) had elevated bycatch per unit effort inside the MPA. All other months had relatively equivalent bycatch per unit effort across boundary lines.

To try to understand monthly variation in bycatch, we consider the seasonality of both olive ridleys and target species. Non-normalized proportions had high bycatch in July and September. July can likely be explained, as this is mid snapper season and an active gillnetting month, with lots of observations. When the data is normalized, July bycatch decreases and October bycatch increases, with September continuing to account for a large proportion in both non-normalized and normalized data. According to small scale fishery landings from 2019 along the Pacific coast of Nicaragua, September is mid-season for snook and lobster fishing, and low-season for snapper (Fig. 10) (División de Planificación, 2019). Of the twenty-one turtle bycatch events observed in September, twenty of them occurred while fishing for snook and one occurred while fishing for snapper. There were no observations of lobster fishing in September.

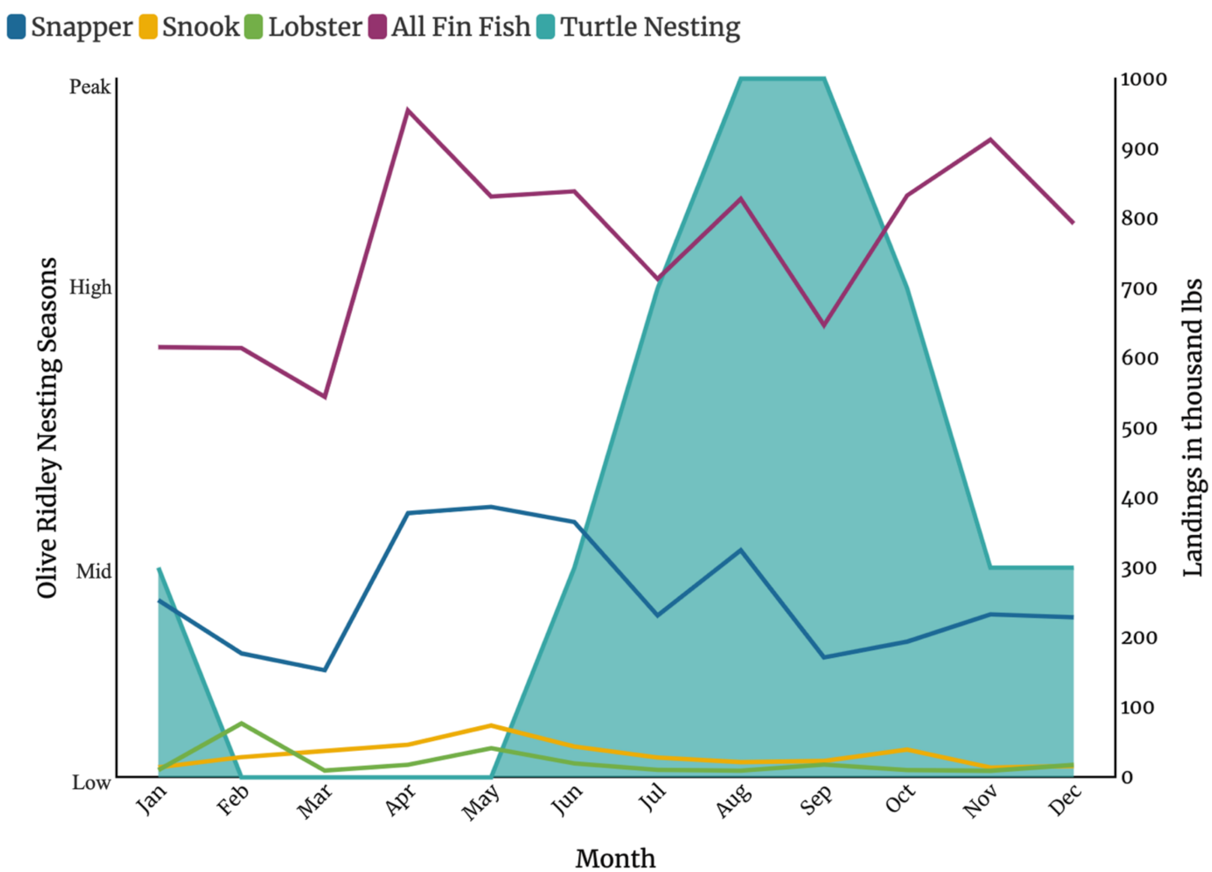

Fig. 10. Annual olive ridley arribada nesting seasons (filled area) in the Eastern Pacific Ocean (Barrientos-Muñoz et al., 2014). Small scale fishery landings (lines) from the Pacific coast of Nicaragua 2019 (División de Planificación, 2019). Figure depicts all small scale fishing activities, not only gillnetting. Note that August and October experience lull in gillnetting, yet longlining continues, contributing to the spike in landings.

There does not seem to be a direct relationship between peak fishing seasons and turtle bycatch. What is clear is that there is nuance across turtle nesting months, both regarding fishing efforts and fisher-turtle interactions. The nuance may be a function of variation in fishing strategies (i.e., location, mesh size) and sea turtle abundance from month to month. Fishers stated that fishing is best when it is raining, yet not storming, and when the water is not too hot. In accordance, there is a lull in gillnetting in August, as an annual dry spell hits in the first half of the month. Fishers continue to target snapper in August, but they generally do so further offshore with longline gear or go south to the border of Costa Rica if they have access to do so, as offshore and/or southern waters are cooler and preferred by the fish. Another lull comes in October, as this is tropical storm season with heavy winds, in which gillnets become thrashed and tangled. In this time, fishers will either simply fish less or continue with alternative gear. High turtle nesting season is from July through October, with peak nesting season being in August and September. Two (i.e., August and October) of the high nesting months experience a lull in gillnetting due to annual weather events. This leaves September as a time that experiences both peak turtle abundance and steady – although not peak – gillnetting.

3.5 Discussion

Overall and based on these observations, the unintentional assumptions underlying the current MPA design and regulations do not appear to hold. It seemed to be unintentionally assumed that with an MPA, gillnetting and turtle bycatch near the nesting site will decline. We found that boundaries are poorly minded by fishers and that only relatively more bycatch occurs inside the MPA. It seemed to be unintentionally assumed that if gillnets are banned, alternative gear will be adopted. We found that half of the fishing fleet uses gillnets. It seemed to be unintentionally assumed that gillnetting for any species is threatening to olive ridleys. We found gillnetting for snook to account for a high proportion of bycatch and believe snook fishing should be studied with regard to its unique use of a large mesh size. And finally, it seemed to be unintentionally assumed that gillnetting during any arribada nesting month is threatening to olive ridleys. We found gillnetting in September to account for a high proportion of bycatch and believe this month should be studied with regard to it being a peak olive ridley nesting.

As such, to answer our broad research questions, the design logics embedded in the MPA management strategies regarding small scale fisheries have not necessarily produced the behavior or outcomes desired. It appears the Chacocente MPA can be better tailored to meet the needs of both olive ridleys and fishers and the following section will explore potential opportunities for doing so based on our findings.

This study contributes to the knowledge of what is happening in and around the Chacocente MPA – an area that has received little at-sea management enforcement and research attention (Hope, 2002; SINAP, 2008). It reveals that management on paper and actual practices are misaligned given the empirical evidence reported here and generates baseline data regarding the interaction between olive ridleys and fishers and a quantification of turtle bycatch. These insights provide new perspectives on the challenges faced by sea turtles, fishers, and conservation and can inform new opportunities.

3.5.1 Implications by extrapolation for turtle bycatch more broadly

El Astillero was chosen for this study due to its proximity to the Chacocente arribada rookery and for being identified by management as one of seven communities of concern regarding uncontrolled fishing. In noting multiple fishing communities of concern, the problem of bycatch is very likely not isolated to El Astillero. To try to understand the potential scale of sea turtle bycatch beyond the 14 fishers we observed in El Astillero, and for the whole Pacific coast of Nicaragua, we conduct an extrapolation with clear assumptions. In Nicaragua, olive ridley protection and management is focused on nesting beaches and eggs. Our study is on adult olive ridleys at sea. In order to grasp the potential scale and impact of bycatch by way of a more commonly used unit (i.e., eggs) in management and conservation circles, we perform an extrapolation and convert mortality of adults via bycatch into eggs lost.

According to INPESCA reports, in 2019 there were 5,759 artisanal fishers on the Pacific coast of Nicaragua (División de Planificación, 2019). According to our field observations, an average of five fishers worked in every one fishing boat. Assuming this same ratio, then there are 1,152 active fishing boats along the Pacific Coast. Assuming the same ratio of gear use as observed in El Astillero, then half of these fishing boats primarily use gillnets, amounting to 576 gillnetting boats. Assuming all gillnetting boats experience the same average number of bycatch events as observed in El Astillero (i.e., 3.6 bycatch events per boat), then an order of magnitude of $2x10^3$ olive ridley bycatch events occurred within the gillnet fleet on the Pacific Coast in 2019. Assuming the same mortality rate as observed in El Astillero (i.e., 79%), then an order of magnitude of $1x10^3$ of the incidentally caught olive ridley turtles died.

For global context, Wallace et al. (2010) found records of 85,000 turtle bycatch events over 18 years. This is for all sea turtle species, and within gillnet, longline, and trawl fishing fleets. These reports were from a very small sample, at <1% of total fleets, and the authors estimate global sea turtle bycatch to be two orders of magnitude larger. For comparison, our sample of 14 gillnetting boats, accounts for 2.4% of the estimated gillnet fleet along the Pacific coast of Nicaragua.

Continuing with the extrapolation, to determine the potential impact bycatch might have on the population, we must examine the proportion of bycatch of mature females (Dapp et al., 2013; Wallace et al., 2010; Whiting et al., 2007). Within the observed sample of olive ridley bycatch from El Astillero in 2019, 51% of turtles were mature females (i.e., curved carapace length is between 65 and 75.2 cm) and of this subsample, 75% were found dead at the time the gillnets were retrieved (Whiting et al., 2007). If we assume the same sex and maturation ratios and mortality rate, then of the estimated bycatch events along the Pacific, approximately 1,000 were of mature females, and of them, approximately 800 died. Mature females lay an average of 2.5 nests per year, with an average of 100 eggs per nest (Barrientos-Muñoz et al., 2014; Bernardo & Plotkin, 2007; Marine Turtle Specialist Group, 2008; Plotkin et al., 1997). As such, each mature female produces 250 eggs annually. Given our assumptions, there was a total loss on the order of magnitude of $2x10^5$ olive ridley eggs, along the Pacific coast of Nicaragua in 2019 through bycatch in gillnets.

For local context, the conservation NGO Casa Congo has saved 25,000 olive ridley eggs over three years via an egg nursery program (Casa Congo, 2020). While it is important that efforts remain to protect eggs on-shore and to curb wildlife trafficking, it is also critical to recognize the unintentional loss occurring at-sea, before the eggs are laid and at risk of trade. This is especially so given limited resources for management and thus the need to identify root problems and prioritize strategies according to population-level impacts (Gerber & Heppell, 2004; Murdoch et al., 2007). Furthermore, the olive ridley has low survivorship of hatchlings (i.e., 1 to 10%), with high survivorship of adults and late reproductive maturity (i.e., mature at 13 years, and live for up to 60 years) (Marine Turtle Specialist Group, 1995; Plotkin et al., 1997). The removal of mature females from the population via bycatch could be having a greater impact on the population than the removal of eggs, depending on the scale at which these threats are occurring (Arlidge et al., 2020; Frazier et al., 2007; Hope, 2002).

This extrapolation makes several assumptions worthy of clarification. Most notably, it assumes that the observations from El Astillero are representative of the entire Pacific coast of Nicaragua. Given ecological, socioeconomic, and fisher strategy variations along the coast, along with the small size of our sample, we cannot say with confidence that our sample is representative beyond the fishers observed in our program (Babcock & Pikitch, 2003). This is the limitation of fisher-dependent data (Ducharme-Barth et al., 2022; Wallace et al., 2010). Further, it is likely that the frequency of bycatch events per gillnetting boat varies along the coast given proximity to the arribada beach and the high abundance of olive ridleys in these areas. That said, it must also be noted that there is a second arribada site in Nicaragua, called La Flor, located to the south of Chacocente. Even if bycatch rates decrease when moving away from Chacocente, they may increase again when moving towards La Flor. Additionally, our spatial data showed that bycatch events do occur outside of the immediate offshore area from the arribada beach. Literature also suggests that distance from shore and depth of water may be determinants of bycatch, given that such variables determine the growth of seagrasses and existence of rocky substrates, and subsequently make up the feeding grounds of olive ridleys (Dawson et al., 2017; Swimmer et al., 2006).

There is much still to learn about the migratory and feedings patterns of olive ridleys and the extent to which bycatch and mortality rates vary across the Pacific coast of Nicaragua. This extrapolation provides a rough consideration of the scale of the bycatch problem regionally. Based on the results produced from the empirical data collected in the observer program in El Astillero and fine analysis of the Chacocente MPA management, much more can be discussed specifically regarding bycatch and fisher interactions at a local level, and so follows.

3.5.2 Limitations of the MPA: A paper park

The Chacocente MPA boundary does not appear to be observed by fishers or enforced in practice, even though it exists as protected on paper. Without historic data of gillnetting activity prior to the establishment of the MPA, we cannot state whether creation of boundary lines and regulations has led to a decline in fishing near the nesting beach. Regardless, the presence of 43% of gillnetting activity taking place within the Chacocente MPA boundaries signifies a need for better communication of information from conservation and governance officials to communities and fishers. Most fishers do not have GPS units and rely on physical geographical features to navigate across the seascape. Fishers commented that confusion does exist over where the boundary line is, with some understanding it to align with the Escalante River. Under this assumption, all of El Astillero Bay would be outside the MPA. This is not the case on paper, as El Astillero Bay is indeed inside the MPA coordinates, with the southern edge of the bay forming the southern border of the MPA. Boundaries could be clearly demarcated at sea and made to align with physical features, enabling an intuitive understanding and communication of boundaries (Center for Behavior and the Environment, 2021).

3.5.3 Strengths of the MPA: In the right place

Given that there was relatively less fishing inside the MPA, yet an equivalent amount of bycatch, suggests that perhaps the boundary line is well placed and that it includes an area of high turtle density. Limiting fishing inside the boundary might in fact decrease bycatch by a larger proportion. However, even if the technical boundary was held in practice and fisher-turtle interactions were eliminated within the MPA, the problem would not be entirely solved with half of bycatch events still occurring outside the MPA. As sea turtles pass through outside waters to get inside the MPA, risk of interaction and bycatch persists (Plotkin, 2010).

3.5.4 Recommendations: What might improve this MPA?

The boundary lines today remain the same as those established in 1983 and the updated Management Plan of 2008 recognized that the boundaries should be redrawn to better align with functions of the ecosystems (SINAP, 2008). We would add that they should also consider socioeconomic systems and areas of importance to community members (Cinner, 2007; Mangubhai et al., 2015). Ultimately, MPA boundaries need to be informed by community and scientific knowledge, and backed by those who engage with and use the MPA and surrounding area and directly influence and are influenced by its effectiveness (Cinner, 2007; Hilborn et al., 2004; Solis Rivera, 2012). Information regarding what the desired behavior is within the MPA and why it matters needs to be communicated – and it needs to resonate with the communities who will either benefit or lose depending on the zoning structures and design (Bennett et al., 2019; Center for Behavior and the Environment, 2021; Mangubhai et al., 2015). Clarity on the boundaries is an imperative step towards good governance. However, it only addresses fishing and bycatch issues occurring within the MPA. To address the problem at large, we must consider recommendations beyond the already implemented, yet questionably effective, conservation intervention.

We believe gillnetting remains high in El Astillero because of its ease as a point of entry to the fishery. Limited financial options available to small scale fishers in Nicaragua makes access to alternative, more costly and involved (i.e., requiring bait or gas to go further off shore) gear not always a feasible option (LaVanchy et al., 2020). If it is decided, through an informed and participatory process, that gillnets should indeed be banned, then certain schemes need to be set in place to adequately compensate and support fishers to turn in gillnets and turn to alternative gear or alternative livelihoods altogether.

Such schemes and support for gillnet alternatives have been tried, with some promising examples. Belizean fishers initially led the movement against gillnets, which the government then supported via a nation-wide ban on gillnets in 2020. The government and various NGOs have set up programs for gillnetters to transition to other forms of income generation (Chanona, 2021). While the schemes appear to be successfully raising funds and offering support to fishers, more time and studies are needed to measure the ecological and socioeconomic effects. Additionally, in Baja California Sur, fishers demonstrated that moving to hook and line gear in place of gillnets was profitable given the selectivity of fishing and better quality of catch, and effective in reducing turtle bycatch (Peckham et al., 2008).

As an alternative to bans, gear modification can be considered. Placing light emitting diodes (LEDs) on the float line of gillnets has been shown to decrease bycatch of green sea turtles by 40% with no significant impact on fish catch (Wang et al., 2010, 2013). Removing the floats of gillnets has been shown to decrease bycatch of loggerhead sea turtles by 68% with no change to fish catch (Peckham et al., 2016). These bycatch reduction technologies are encouraging, yet would require testing with olive ridley species and in the Pacific coast of Nicaragua specifically to determine if such solutions are viable for this case (Senko et al., 2014).

The fate of gillnets, as a ban or with modification, needs to be informed by and co-designed with fishers to be successful (Jenkins, 2010; Senko et al., 2014). The level of social capital within a community and trust in those who are initiating the uptake of new practices (i.e. government, NGO, community members) will likely play a role in determining fishers willingness to adopt or not (Digal & Placencia, 2017). In studying turtle exclusion devices, another type of bycatch reduction technology designed for shrimp trawlers, Jenkins (2010) found that fishers invented the most successful devices that effectively met conservation needs and limited effect on fishing. Given fisher's knowledge of the problem and incentive for a solution, they are aptly positioned for designing and leading mitigation strategies. Furthermore, the "Local Inventor Effect," reveals a disproportionately high rate of adoption with geographical proximity to where the invention was created (Jenkins, 2010). This can likely be attributed to degrees of trust, as the uptake of sustainable practices has been shown to vary in accordance to the dynamics between individuals and community, and between individuals and institutions (Falk & Guenther, 1999).

Species and seasonal closures are other management tools that appear to be applicable in the case of Chacocente given the variation in turtle bycatch between targeted species and between months. Closing the snook gillnet fishery specifically could address the largest proportion of turtle bycatch, while allowing the snapper and lobster gillnet fisheries to remain open would enable fishers to still have access to valuable catch, which is critically important for livelihood security. Snook appears to offer relatively small economic contribution in comparison to snapper on both a local and national level. However, more work should be done to understand the exact extent to which fishers depend on the snook fishery and the non-economic value it holds within the community (Teh et al., 2015).

The olive ridley nesting season is high from July through October, with peaks in August and September. While turtle bycatch was relatively low in August, it was high in September. All but one of the turtle bycatch events in September occurred while gillnetting for snook. We infer that the high abundance of nesting turtles, combined with fair conditions that enable gillnetting for snook, may be drivers of high bycatch in this month. Based on these initial observations and inferences, the month of September could be considered for a seasonal closure to address the largest proportion of turtle bycatch. More work is needed to know what exactly is occurring ecologically and socially at this time in relation to turtle bycatch, and to understand what the impacts might be if a monthly closure was put in place.

With a lack of alternative livelihoods in El Astillero and the Chacocente region as a whole, any species or seasonal closures may result in increased fishing pressure on other species at different times (LaVanchy et al., 2020; Senko et al., 2014). If the targeted species are at or under maximum sustainable yield, than such pressure may be acceptable (Pauly & Froese, 2021; Sumaila & Hannesson, 2010). However, there are few studies on the state of exploitation in the Pacific coast of Nicaragua. INPESCA has noted a decrease in certain species landings and an increase in small scale fishing fleets, which remain open access (Navarro, 2010). With these observed trends, tradeoffs and outcomes from closures would need to be anticipated and monitored.

3.5.5 Governance: Support for fishing communities

Feasibility of enforcement of boundaries, gear types, and closures must also be considered. The Management Plan of 2008 recognized the lack of, and need for, at-sea monitoring, along with the difficulty in doing so due to limited financial resources. In 2011, the International Collective in

Support of Fishworkers conducted interviews with fishers in El Astillero, who echoed frustration over unsustainable and illegal fishing activities taking place without ramification (Solis Rivera, 2012). Based on field observations in 2019, these gaps in enforcement appear to remain. The shared desire for better enforcement by both management and fishers is a low hanging fruit for collaboration and co-management. The Nicaraguan Ministry for Environment and Natural Resources (MARENA), National System of Protected Areas (SINAP), and Nicaraguan Institute of Fisheries and Aquaculture (INPESCA) have all made written calls to move towards decentralized governance and community participation, yet there is a lack of structure to do so and the voices of small scale fishers have not been included in governance processes (Cotto & Marttín, 2007; SINAP, 2008; Solis Rivera, 2012). Additionally, the non-governmental organization, Casa Congo, established in 2017 and run in international and local partnership, has established relationships with both MARENA and the community of El Astillero and works toward sustainable community development. Their presence as a third party may have a role to play in collaboration and governance to meet the objectives of both fishers and resource managers.

Nicaragua continues to govern from a top-down approach, as seen at a national level in the current authoritarian government, on a regional level with fishers not being informed regarding the occurrence of fisher association meetings, and on a local level through the use of militarized conservation on the Chacocente nesting beach (Crawford et al., 2010; Gonda et al., 2022; Haas et al., 2015; Jentoft & Midré, 2011; Martí i Puig & Serra, 2020). This discrepancy again between paper and practice shows good intention, yet an inability to follow through with implementation and one must question why, especially given that Nicaragua is a tumultuous political climate both historically and at present. Is the inability of the government to decentralize simply due to the difficulty in organizing across institutions and levels within the sector, or are there active barriers imposed out of an unwillingness to relinquish power over resources?

Regardless, communities are not powerless. While cooperation between government and community is ultimately critical for community-based marine resource management, it can be initiated without government support (Pomeroy & Berkes, 1997). The body of literature on common pool resources clearly shows that resource users can, and do, self-organize to form management systems that move towards sustainable social-ecological systems (Ostrom, 2009). For example, in the case of Maine's lobster fishery, fishers abide by certain regulations not because they are heavily enforced by government officials, but because they are accepted as social norms within the community and to break the regulation would mean separation from the greater social fabric (Acheson, 2011). Norms and values are set within communities, so while members of El Astillero are not outwardly invited to participate in management decisions, they do have autonomy to define their own principles within the fishery at the local level (Ross, 2015; Sollis, 1989).

If fishers were to self-organize within their own social context, they could next seek support from respectable organizations, such as international universities, conservation organizations, and funding agencies, to further back their agreed upon approaches of organization and resource use. Eventually, the government could see this national and/or global support as a positive and wish to stake a claim in the benefits that come from it, further leading to possibilities for cooperation and true co-management. This is likely a long and strategic process, yet it is possible and worth embarking upon, for once arrangements at the national and community level are set in place, resources would likely be better managed and rights distributed more equally, further ensuring a greater sense of livelihood security for fishers in the long run (Pomeroy, 1995; Pomeroy & Berkes, 1997; Romano, 2017).

3.6 Conclusion

On-shore management of the Chacocente arribada site is part of, and influenced by, a long historical, cultural and political legacy as concerns olive ridley turtle eggs. This burden may cross over to at-sea management of the fisheries, given that the involved stakeholders are largely the same. However, there is little history of management or contention over resources at-sea, and there is a seemingly shared objective to decrease sea turtle bycatch (Campbell, 2007; Faber, 1993; Hope, 2002; SINAP, 2008). As such, this problem may also pose an opportunity for building a bridge between stake holders.

Turtle bycatch is a challenge that both conservationists and fishers in this area recognize and want to address, although the scale of this problem has been largely unstudied and unrecorded. Due to a focus on olive ridley egg conservation on-shore, difficulties of at-sea management and enforcement, and a lack of incorporation of the small scale fishing sector in governance, the problem has remained (Hope, 2002; SINAP, 2008; Solis Rivera, 2012). The aforementioned recommendations based on our observation would not eliminate bycatch as a whole. However, the results of this study do reveal windows of opportunity and potential solution points of entry for changing the magnitude of the turtle bycatch problem.

The empirical data of this study is based on one pilot field season, with a small sample size using convenient sampling, which presents limitations. According to fisheries literature, sea turtle bycatch is considered a “rare” event, given that the species is less abundant and its frequency of catch is minimal, relative to the abundance and catch of targeted species (Babcock & Pikitch, 2003; Lewison et al., 2014). When rare bycatch events occur within small samples (i.e., few observations of fishing effort), the rate of bycatch becomes exacerbated. As such, there is debate within the literature of how much observer coverage is enough to estimate bycatch. Babcock and Pikitch (2003) suggest, "coverage levels of at least 20 percent for common species, and 50 percent for rare species," to construct, "reasonably good estimates of total bycatch." We covered 46 percent of the gillnetting fleet of El Astillero in 2019. Within our sample, our results showed a highly elevated bycatch per unit effort inside the MPA during the month of September. We believe this is due to particularly few observations of fishing effort inside the MPA within this month, and a subsequently skewed rate of bycatch. While we are confident in our coverage of September, we are less confident in comparison of bycatch rates inside and outside of the MPA boundaries in this month, simply because the majority of the observed fishing effort took place outside the MPA.

Accurate knowledge and documentation of behavior at-sea is a major challenge for fisheries management and marine conservation (Pomeroy, 1995; Pomeroy & Berkes, 1997). Observer programs are one of the most reliable ways to document fisher behavior, however, the sampling process is complex to implement and poses limitations (Babcock & Pikitch, 2003). Our data carries bias, through non-random sampling and by the potentially altered behaviors of fishers in the presence of an observer (Babcock & Pikitch, 2003; Brooke, 2015). Furthermore, while all observers underwent the same training, there may be variation in their data collection methods and reporting. We cannot speak to turtle behavior apart from fishing behavior, which is the limited nature of fisher-dependent data (Barange et al., 2014). Nonetheless, these preliminary results reveal initial gaps and patterns that can be of value to future research, especially given that olive ridley bycatch has not been previously quantified in this region (Marine Turtle Specialist Group, 2008; Wallace et al., 2010).

Our study raises more questions and highlights details that should be considered in the design of the Chacocente MPA fishing regulations. Given the preliminary nature of this study, more understanding of turtle and fisher behavior in this area is needed. The extensive knowledge of El Astillero fishers who have worked in and observed the Chacocente area for decades should be recognized, and their voices and priorities at the forefront of MPA design (Eriksson et al., 2016; Solis Rivera, 2012). The integration of fishers is important in Nicaragua, and also globally where coastal communities and MPAs overlap (Jentoft, 1989, 2000; Kalikoski & Satterfield, 2004; Pomeroy, 1995).

For us to move towards the aspired 30% marine protection by 2030 in practice – and beyond mere paper parks – it is imperative that we meet the needs of both marine life and coastal community livelihoods (Bennett & Dearden, 2014; Cinner, 2007; Edgar et al., 2014; Mangubhai et al., 2015; Sala et al., 2021). Doing so requires acknowledgement and understanding of the unique ecological and socioeconomic dynamics at play, and subsequent design of management not based on assumption, but rather on examined and experienced behaviors and relationships.

Chapter 4: Conclusion

Having contextualized the history of the Chacocente refuge and revealed flawed, unintentional assumptions embedded in the marine protected area regulations, I wish to close with a few reflections on future approaches to management design. Management of the Chacocente Rio Escalante Wildlife Refuge has historically left fishers out of the conversation (Solis Rivera, 2012). Yet since fishers are the ones most directly impacted by MPA management decisions (e.g., resource access gains and losses), knowledgeable of oceanic challenges, and in situ to practice solutions, it is paramount that they be considered in governance processes as stake holders (Béné, 2003; Jenkins, 2010; Teh et al., 2015). In part, success of MPAs depends upon fisher perceptions and the degree to which their needs and aspirations are met as beneficiaries (Bennett et al., 2019; Buglass et al., 2018; Cinner, 2007; Edgar et al., 2014). Bennett et al. (2019) found support from small scale fishers for MPAs specifically, and conservation in general, was strongly associated with perceptions of good governance (e.g., communication of information, transparency in decision-making, participation and voice, trust, rule of law) and social impacts (e.g. livelihoods, food security, knowledge and education, community social well-being).

In our case study, we did not measure community perceptions and associated support for the Chacocente MPA, and so cannot speak to this in full. Though, from reports and field observations, there are apparent areas of lack and opportunities for improvement regarding engagement and perceptions. With ambiguity regarding when the MPA fishing closure is in existence and where the borders lie, there appears to be a lack of communication of information. With inadequate notifications of fisher association meetings, there appears to be a lack of transparency. With continued top-down governance and no support for fisher involvement, there appears to be a lack of participation (Solis Rivera, 2012). With the legacy burden from militarized management strategies on the Chacocente beach regarding olive ridley eggs, there appears to be a lack of trust (Campbell, 2007). With inconsistency between intensive regulation on-shore and absence of regulation at-sea, there appears to be a lack of rule of law (Campbell, 2007; Hope, 2002; LaVanchy et al., 2020). Since the ban of turtle products in 2005, there appears to be a loss of livelihood and food supplement via turtle eggs (Campbell, 2007). Furthermore, with the expressed perception of the Chacocente refuge as "dangerous," its existence does not appear to contribute to community social wellbeing.

In recognizing these deficiencies, and in consideration of Chacocente MPA design, we suggest using an integrated human centered design and systems thinking approach (Fig. 11) (Both, 2018; Laszlo & Laszlo, 1997). Together, they enable a wide lens to see complex systems as a whole and the relationships between the parts, while keeping in mind motivations of, and impact on, the behavior of stake holders (Béné et al., 2016; Brown, 2008; Hall & Fagen, 1968; Senge, 1990). We also suggest incorporating a behavior centered design approach, which recognizes six main categories of intervention, including: social influences, information, emotional appeals, rules and regulations, choice architecture, and material incentives (Center for Behavior and the Environment, 2021). These categories largely overlap with what Bennett et al. (2019) identified as influencing support for conservation and make a promising suite of levers to explore within the case of Chacocente. MPAs involve complex social-ecological systems, with no one-size-fits-all management strategy (Chan et al., 2012; Edgar et al., 2014; Gill et al., 2019; Mangubhai et al., 2015). By incorporating behavioral science, system science, and design thinking into resource management, there is potential to tailor governance for a target audience and behavior, and impart focused leverage on specific environmental challenges (Bennett et al., 2019; Brown, 2008; Center for Behavior and the Environment, 2021; Ostrom, 2009).

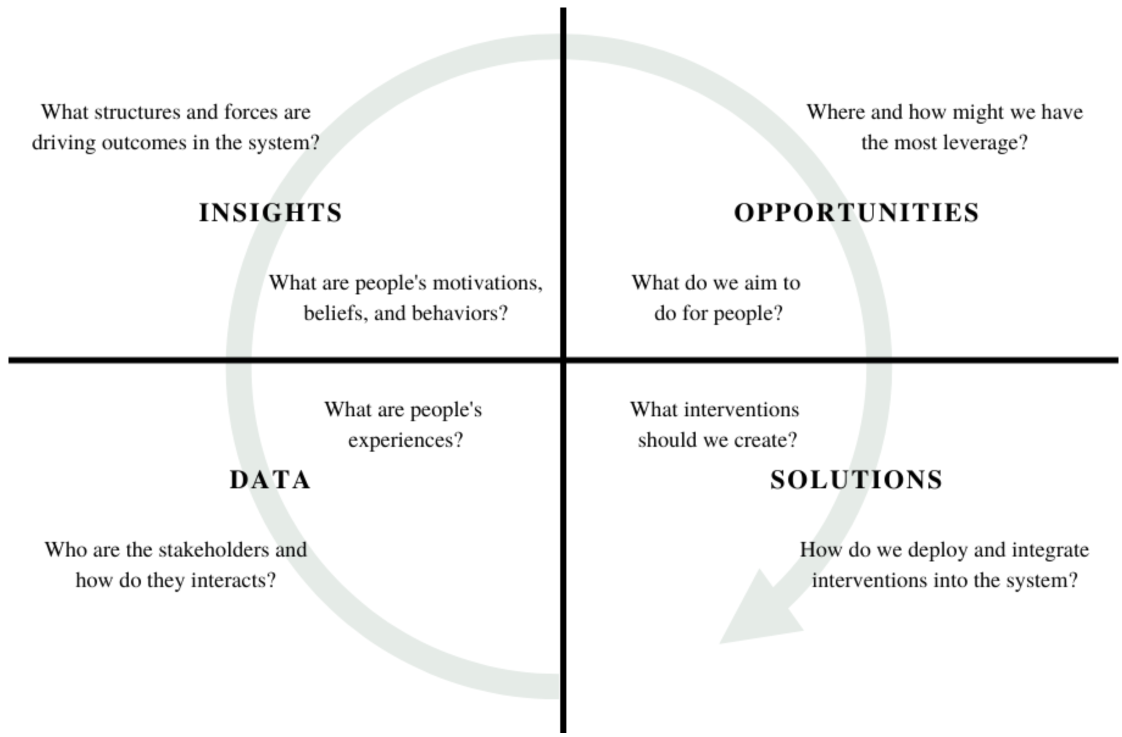

Fig. 11. Questions outlining a human-centered and system-thinking design approach. Adapted from Both, 2018.

In our case study, the target audience is fishers of El Astillero, and target behavior is unintentional catch of olive ridleys. In viewing this behavior as part of a larger system, it is critical to consider other actors, rights, and stake holders, their roles and relationships, and how they might be influencing fisher-turtle interactions. For example, actors may include fishers from other communities, fish buyers, and governing officials from MARENA and INPESCA. Questions of relationality may include: how might the behavior of fishers in other communities influence the decision of fishers from El Astillero to fish near the Chacocente arribada site; how might market structures with fish buyers influence decisions to join the snook gillnet fishery; how might support from governing officials affect the willingness of El Astillero fishers to prototype a bycatch solution?

According to the Chacocente Management Plan the, “main actors by right, obligation, responsibility and authority in management, include: private owners (84 private owners have been recorded), MARENA, and the Municipal Governments,” (SINAP, 2008). It is unclear if El Astillero is inside or outside the refuge boundary and subsequently unclear regarding community member rights. In written documentation, El Astillero is not included on the list of communities inside the refuge, yet according to maps and coordinates, half of the town is within the boundary lines. This is important to clarify, because if land owners in El Astillero are technically private owners in Chacocente, they would have a right and authority to inclusion in management (SINAP, 2008). This would give further grounds for representation in decision making and could also change community perceptions as beneficiaries of the refuge and subsequently influence behavior.

This study is the first stage of a long term project. From primary and secondary sources, we gathered data and analyzed insights on current challenges, and discussed opportunities to better meet the Chacocente refuge aim of “productive conservation.” We plan to share the insights and opportunities with El Astillero fishers and other identified actors within and around the Chacocente refuge. Additionally, we will adjust and bolster the understanding of this case with any lived experiences and knowledge they choose to share. Through an iterative and collaborative approach, adoptable solutions can then be identified and designed (Brown, 2008; Center for Behavior and the Environment, 2021; Jenkins, 2010; Teh et al., 2015). In the words of Jacqueline Novogratz (2020), “When we fail to listen to those the world excludes, we lose the possibility of solving problems that matter most to all of us. But when we succeed at listening with all our attention and empathy, we have a chance to set others and ourselves free.” In the case of El Astillero fishers and the Chacocente Rio Escalante Wildlife Refuge, we aim to continue listening.

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Appendices

Appendix A. Permissions of Chacocente Rio Escalante Wildlife Refuge

Table A.1. Marine zoning and fishing regulations within Chacocente MPA (SINAP, 2008).

Zone	Definition	Objective	Permitted	Not Permitted
Marine conservation zone, subzone of conservation of marine resources	Borders the Coastal Conservation zone to the east and extends for 5 nautical miles to the west. Depth ranges from 2 to 30 m, with rocky outcrops. Rich in marine diversity, both vertebrate and invertebrate organisms.	Regulate small scale and industrial fishing in such a way as to guarantee long-term ecological and economic sustainability of the activity.	Small scale fishing under the vertical bottom longline modality, bottom hand line with single or multiple hook and fishing rod with hook, valid for the communities surrounding the refuge. Sport fishing under the modality of catch and release of the specimens. Practice recreational or contemplation diving, as well as dives for scientific purposes or for monitoring biodiversity.	Small scale fishing under the modalities of setting pots for fish and lobsters, laying of surface, drift, mid-water or bottom gillnets, beach seines and any fishing method that involves dragging fishing equipment on the bottom, to midwater or shallow. Using fishing gear that affects marine biodiversity or that causes the death of captured fish, such as setting pots for fish and lobsters, laying gillnets drifting or midwater or bottom, beach seines and any fishing method that involves dragging fishing equipment on the bottom, midwater or superficial. Carry out extractions of organisms or parts of them, both invertebrates and vertebrates, and collect their eggs or young.

Zone	Definition	Objective	Permitted	Not Permitted
Marine conservation zone, subzone of migratory marine species protection	Borders the marine resources sub-zone to the east and extends to the limit of the Nicaraguan territorial sea to the west, 12 nautical miles from shore.	Protection of important habitats for the migration of species of international importance and the reproduction and feeding of sea turtles.	Small scale fishing under the vertical bottom longline modality, bottom hand line with single or multiple hook and fishing rod with hook, valid at communities surrounding the refuge.	Small scale fishing under the modalities of setting pots for fish and lobsters, laying surface or drifting gillnets, and any fishing method that implies dragging fishing equipment on the bottom, to midwater or shallow.
	Depth is 40 m and greater, with rocky reef on bottom.		Practice recreational or contemplation diving, as well as dives for scientific or biodiversity monitoring purposes.	Extract sea turtles and their eggs; as well as sharks, whales, dolphins and rays or any other species in national closure, in the CITES appendices or in the IUCN red lists.
	Passage area for migratory marine species of international importance, and important site for mating and feeding of sea turtles prior to spawning on the beaches of the refuge.			Industrially extract lobsters, shrimp or crabs and invertebrates

Appendix B. Per Unit Effort Calculations

Table B.1. Raw data and calculated bycatch (BPUE), catch (CPUE), and value (VPUE) per unit effort for all fishing observations, inside and outside of the MPA boundaries. Note spatial data was not always collected due to technological difficulties. When spatial data not available, orientation to MPA boundaries was inconclusive and so not differentiated as 'inside' or 'outside', yet still included in 'overall' totals.

	Net Length (m)	Turtles Captured (#)	Fish Captured (lbs.)	Value ($ USD)	BPUE	CPUE	VPUE
Overall	138310	50	6871	3487	0.04	4.97	2.52
Inside MPA	54320	24	2478	1284	0.04	4.56	2.36
Outside MPA	71310	24	3993	1988	0.03	5.60	2.79

Table B.2. Raw data and calculated bycatch (BPUE), catch (CPUE), and value (VPUE) per unit effort when targeting species, inside and outside of MPA. Data from observer program, 2019. Note spatial data was not always collected due to technological difficulties. When spatial data not available, orientation to MPA boundaries was inconclusive and so not differentiated as 'inside' or 'outside', yet still included in 'overall' totals.

Species	Net Length (m)	Turtles Captured (#)	Fish Captured (lbs.)	Value ($ USD)	BPUE	CPUE	VPUE
Snapper Overall	75690	6	4361	2398	0.01	5.76	3.17
Snapper In	19880	1	1074	667	0.01	5.40	3.35
Snapper Out	45050	3	2891	1516	0.01	6.42	3.37
Snook Overall	18760	27	1243	575	0.14	6.63	3.06
Snook In	10120	12	805	250	0.12	7.95	2.48
Snook Out	8640	15	438	324	0.17	5.07	3.75
Lobster Overall	38460	14	844	450	0.04	2.20	1.17
Lobster In	19120	9	181	302	0.05	0.94	1.58
Lobster Out	17420	5	660	148	0.03	3.79	0.85

Table B.3. Raw data and calculated bycatch (BPUE), catch (CPUE), and value (VPUE) per unit effort when fishing in each month, inside and outside of MPA. Data from observer program, 2019. Note spatial data was not always collected due to technological difficulties. When spatial data not available, orientation to MPA boundaries was inconclusive and so not differentiated as 'inside' or 'outside', yet still included in 'overall' totals.

Month	Net Length (m)	Turtles Captured (#)	Fish Captured (lbs.)	Value ($ USD)	BPUE	CPUE	VPUE
June	24840	4	1765	993	0.02	7.10	4.00
June In	7700	1	415	250	0.01	5.39	3.24
June Out	12340	2	1147	596	0.02	9.30	4.83
July	69630	20	2423	1608	0.03	3.48	2.31
July In	34020	9	1071	723	0.03	3.15	2.13
July Out	30690	11	1220	866	0.04	3.97	2.82
August	18980	2	1879	346	0.01	9.90	1.82
August In	7720	1	881	199	0.01	11.41	2.57
August Out	11260	1	998	147	0.01	8.86	1.31
September	11140	21	480	318	0.19	4.31	2.85
September In	1200	11	54	38	0.92	4.50	3.18
September Out	9940	10	426	280	0.10	4.29	2.82
October	6120	2	163	75	0.03	2.67	1.23
October In	1080	2	1	0	0.19	0.09	0.00
October Out	4680	0	154.41	75	0.00	3.30	1.61
November	7600	1	161	146	0.01	2.11	1.92
November In	2600	0	56	74	0.00	2.15	2.84
November Out	2400	0	47	24	0.00	1.96	0.98