

**SEA TURTLE NESTING TRENDS FROM 2011-2017 ON THE
OSA PENINSULA, COSTA RICA**

by

Megan Ossmann

Drs. Matthew Godfrey and Lisa Campbell, Advisors

April 17, 2019

Masters project submitted in partial fulfillment of the
requirements for the Master of Environmental Management degree in
the Nicholas School of the Environment of
Duke University

EXECUTIVE SUMMARY

Sea turtles play an important role in marine ecosystems and are socioeconomically valuable through ecotourism in many parts of the world. Six of the seven species of sea turtles are listed globally as either vulnerable, endangered, or critically endangered, largely due to anthropogenic threats. The nesting and incubation period for sea turtles is particularly vulnerable to threats such as poaching of eggs, introduction of predators, light pollution, and development of beaches.

A large component of sea turtle conservation involves the protection of nesting females, their eggs, and resultant hatchlings on nesting beaches. Sea turtle nesting beaches are often monitored by staff and volunteers at nonprofit organizations, state or federal agencies, and/or researchers in academia. Osa Conservation is a nonprofit organization that monitors two nesting beaches on the Osa Peninsula, Costa Rica, a region that is particularly important for solitary-nesting olive ridley (*Lepidochelys olivacea*) turtles. From 2008 to 2018, Osa Conservation monitored Playa Piro and Playa Pejeperro, collecting data on nest abundance, spatial and temporal distribution of nests, predation, and hatch success. In more recent years, Osa Conservation has relocated nests affected or threatened by erosion or predation to a protected hatchery located on Playa Piro.

There is limited existing literature on marine turtle nesting on the Osa Peninsula so a comprehensive dataset containing information about regional nesting trends is a valuable contribution to research. Using data collected from 2011-2017, this document provides Osa Conservation with a detailed analysis of the two beaches they monitor, as well as data collected from the hatchery from 2015-2017. My objectives were to provide the following information:

- the general and notable trends in nesting behavior, predation, success of emerged nests, biological data collected from adult females, and the hatchery;
- whether these data support previous research and literature about nesting preferences and biological characteristics of these species;
- the key similarities and differences between Playa Piro and Playa Pejeperro;
- and how the data compare to previous research in this same study area and what changes have occurred

The data collected at Playa Piro and Playa Pejeperro throughout this study support literature about nesting preferences and biological data on both the olive ridley and the green turtle (*Chelonia mydas*), the two most abundant sea turtle species in the region. Both beaches are similar in species composition, nesting frequency, hatch success, and predation. The mean hatch success of nests was 7% and the mean predation rate of nests was 22%. The relocation of eggs

to the hatchery at Playa Piro has proved to be a consistent success, with a release rate averaging 86%. The greatest difference between the two beaches is the dominant predator types. Playa Piro experiences greater predation by coatis and raccoons, and Playa Pejeperro experiences greater predation by dogs. On both beaches, the predation rate has been steadily increasing across the study period, nearing 50% of nests in 2017. When compared to previous studies conducted on these beaches, it is apparent that the predation of nests remains a significant threat, though the composition of predators has changed. There is significantly greater predation by coatis and raccoons, and significantly less poaching of eggs by humans than in the past.

With these results of this study, I provide insight and recommendations for Osa Conservation's monitoring and management plans, which may contribute towards their effort to limit threats to sea turtles and increase hatching success in this important region. My recommendations are as follows:

- 1) The continuation of monitoring and data collection in this region;
- 2) Expansion of monitoring to other beaches and/or data sharing among organizations;
- 3) Adoption of State of the World's Sea Turtles (SWOT) protocol;
- 4) More defined predation categories;
- 5) The use of predator excluders to reduce predation;
- 6) The continuation of nest relocation to the hatchery with additional research on its effects

This document summarizes key findings from the data collected by Osa Conservation from 2011-2017. It provides an update to previous monitoring studies conducted in this region, and serves as baseline data with which to compare future studies. This research will help Osa Conservation improve their monitoring and management strategies regarding nesting, and contribute to better understanding of nesting trends in this important region.

TABLE OF CONTENTS

Executive Summary	ii
Table of Contents	iv
List of Tables	v
List of Figures	vi
Introduction	1
Study Area.....	2
Nesting Behavior	3
Threats.....	3
Conservation.....	4
Methods	4
Data Collection.....	4
Data Analysis.....	6
Results	8
Effort.....	8
Nest Characteristics.....	11
Predation.....	15
Live Nesting Females.....	22
Wild Nests.....	28
Hatchery.....	30
Discussion	33
Comparison to Previous Research.....	36
Drake (1993).....	36
Drake (1996)	37
Barquero-Edge (2013)	37
Summary.....	37
Management Recommendations	40
Conclusion	43
Acknowledgements	44
References	45

LIST OF TABLES

Table 1. Predation percentage of nests by each predator type.

Table 2. The mean percent of eggs predated in each nest per year.

Table 3. Minimum, maximum, median, and mean curved carapace length.

Table 4. Minimum, maximum, median, and mean distance to tide.

Table 5. Minimum, maximum, median, and mean distance to vegetation.

Table 6: Comparison of this study to three previous sea turtle nesting studies on the Osa Peninsula.

LIST OF FIGURES

Figure 1: Map of study area.

Figure 2: Effort per year, as measured by number of nights and time in hours spent patrolling.

Figure 3: Effort as measured by the number of patrols per month across the study period.

Figure 4: Effort as measured by the total patrol time (hours) per month across the study period.

Figure 5: Percentage of each type of nest out of total nest count.

Figure 6: Number of nesting events per species, per year.

Figure 7: Seasonal nesting trends.

Figure 8: The frequency of nests within each sector of beach.

Figure 9: The frequency of nests in relation to distance from vegetation (m).

Figure 10: The number of olive ridley and green turtle nests in each beach zone.

Figure 11: Change in predation rate of nests across study period.

Figure 12: Percent of predated nests by humans, dogs, and coati each month (2015-2017).

Figure 13: The percentage of nests predated by each predator type.

Figure 13: Percentage of all predated nests by each predator type per year.

Figure 15: Mean predation percentage and SD of each predator type averaged across all years.

Figure 16: Number of adult nesting females encountered during patrols, per year.

Figure 17: Curved carapace length (CCL) in cm for the olive ridley and green turtle.

Figure 18: Distance to the tide (m) at which nesting females were recorded.

Figure 19: Distance to tide (m) at which nesting females were recorded per year.

Figure 20: Distance to the vegetation (m) at which nesting females were recorded.

Figure 21: Distance to vegetation (m) at which nesting females were recorded per year.

Figure 22: Mean beach width per year.

Figure 23: Hatch success and emergence success of nests per year for wild (hatched) nests.

Figure 14: Seasonal trends in nest success.

Figure 15: Clutch size distribution of nests across all years.

Figure 16: Success rate of hatchery for 2015-2017.

Figure 17: Clutch size distribution of excavated hatchery nests.

Figure 18: Average hatchery success rate by month in 2016.

INTRODUCTION

Sea turtles play an important role in marine ecosystems, such as altering habitat through foraging and serving as a source of prey for predators at a higher trophic level (The National Academies of Sciences, Engineering, and Medicine 2017; Ruckdeschel and Shoop 2006; Heithaus 2013). Sea turtles also have a significant socioeconomic role through ecotourism in many parts of the world (The National Academies of Sciences, Engineering, and Medicine 2017; Ruckdeschel and Shoop 2006). Globally, six of the seven species of sea turtles are listed as either vulnerable, endangered, or critically endangered (Abreu-Grobois and Plotkin 2008; Mortimer and Donnelly 2008; Seminoff 2004; Wallace et al. 2013). This is due to declines in populations that have been largely caused by human activities, such as entanglement in fishing gear and habitat degradation (National Research Council 2010). Sea turtles are also sensitive to anthropogenic threats when nesting on the beach, and while their eggs are incubating, through poaching, introduction of predators, light pollution, and development of beaches (Drake 1996; Korein 2019; The National Academies of Sciences, Engineering, and Medicine, 2017).

A large component of sea turtle conservation involves the reduction of mortality during the nesting, egg incubation, and hatchling emergence period (The National Academies of Sciences, Engineering, and Medicine 2017; SWOT Scientific Advisory Board 2011). Typically, nesting data are collected through nightly patrols on the beaches at which individual turtles nest. Data are often collected through staff and volunteers at nonprofit organizations, state or federal agencies (particularly in the United States), and/or researchers in academia (The National Academies of Sciences, Engineering, and Medicine 2017). In particular, multi-year datasets collected at the same study area allow for the determination of trends within the study period, as well as comparison to earlier studies within the same region (SWOT Scientific Advisory Board 2011). Monitoring patterns and changes in nesting effort over time allows for not only baseline data to be established, but also facilitates the evaluation of the efficacy of any management strategies that have been implemented.

Osa Conservation is a nonprofit organization located on the Osa Peninsula, Costa Rica, that works to protect the globally significant biodiversity of the region (Osa Conservation n.d.). A large component of their conservation work is sea turtle monitoring and patrol of nests, particularly of olive ridleys (*Lepidochelys olivacea*) which is the most abundant species in the region (Drake 1993; Drake 1996; Barquero-Edge 2013). There is limited existing literature on marine turtle nesting on the Osa Peninsula (Drake 1996), so a comprehensive dataset containing information about regional nesting trends is a valuable contribution to our understanding of nesting characteristics in this area. Based on data collected from 2011-2017, I provide Osa Conservation with a detailed analysis of the two beaches they monitor, Playa Piro and Playa Pejeeperro. My objective is to provide the following information:

- the general and notable trends in nesting behavior, predation, success of hatched nests, biological data collected from adult females, and the hatchery;
- whether these data support previous research and literature about nesting preferences and biological characteristics of these species;
- the key similarities and differences between Playa Piro and Playa Pejeperro;
- and how the data compare to previous research in this same study area and what changes have occurred

With the results of this study, I aim to provide insight and recommendations for Osa Conservation’s monitoring and management plans, which may help to improve the effort to limit threats to sea turtles and increase hatching success in this important region.

STUDY AREA

The Osa Peninsula is located on the Southwest Pacific Coast of Costa Rica. It is one of the highest conservation priorities in Central America due to its high levels of biodiversity and endemism (Ankerson et al. 2006). In contrast with other marine turtle nesting beaches in Costa Rica, such as Ostional, Tortuguero, Playa Grande, and Nancite, few studies on sea turtle nesting on the Osa Peninsula have been published (Drake 1996). Osa Conservation conducts monitoring on two adjacent beaches in the region, Playa Piro (2 km) and Playa Pejeperro (4.5 km) (Figure 1). To the west of these beaches is Corcovado National Park, a protected area with one of the highest legal levels of protection in Costa Rica (Ankerson et al. 2006).



Figure 1: Map of study area.

NESTING BEHAVIOR

The beaches in this region are important nesting areas for four species of sea turtles that nest in the Costa Rican Pacific: olive ridleys, green turtles (*Chelonia mydas*), hawksbills (*Eretmochelys imbricata*), and leatherbacks (*Dermochelys coriacea*) (Barquero-Edge 2013, James and Melero 2015). Olive ridleys are considered the most abundant sea turtles in the world (Burger and Gochfeld 2014a,b; Dornfeld et al. 2015; Hart et al. 2014; Polovina et al. 2004), and are the most abundant nesters in this region, followed by green turtles, and hawksbills and leatherbacks which nest infrequently (Korein et al. 2019, Barquero-Edge 2013; Drake 1996). Some of the largest nesting aggregations of olive ridleys in the Eastern Pacific occur in Costa Rica, typically at mass nesting, or “arribadas”, locations such as Playa Nancite and Playa Ostional (National Marine Fisheries Service and U.S. Fish and Wildlife Service 1998; Burger and Gochfeld 2014a; Dornfeld et al. 2015). On Playa Piro and Playa Pejeperro, olive ridleys behave as solitary nesters, in contrast with these mass assemblages (Barquero-Edge 2013). Arribada nesting has been the focus of the majority of previous studies (Dornfeld et al. 2015), despite research that suggests solitary nests have higher hatchling emergence success than arribada nests, making them important contributors to the olive ridley population (Dornfeld et al. 2015; Barquero-Edge 2013).

Mature female olive ridley turtles nest all year in the eastern Pacific, with peak months for arribadas occurring from September through December (NMFS and USFWS 1998). Individuals generally nest every year, laying one to two clutches (Dornfeld et al. 2015). The preferred nesting habitat of this species is a relatively flat area in the middle of the beach that is free of debris (NMFS and USFWS 1998), where they will dig nests and lay clutches of about 100 eggs (NMFS and USFWS 1998, Hart et al. 2015). The adults will return to the ocean, and the eggs will incubate for 50 to 60 days before they hatch (NMFS and USFWS 1998).

THREATS

According to the IUCN Red List, olive ridleys and leatherbacks are globally listed as vulnerable, green turtles are globally listed as endangered, and hawksbills are globally listed as critically endangered (Abreu-Grobois and Plotkin 2008; Mortimer and Donnelly 2008; Seminoff 2004; Wallace et al. 2013). Due to long lifespan and wide-ranging migration, sea turtles are threatened at all life stages (National Research Council 2010). They are susceptible to numerous threats during the incubation period: poaching and overharvesting of eggs by humans; nest predation; erosion of nests; development along nesting beaches; and long-term environmental effects due to climate change, such as sea level rise and influence on nest temperature (Barquero-Edge 2013, Drake 1996, Burger et al. 2014). Once the turtles emerge from eggs and reach the ocean, they face numerous threats such as accidental capture in fishing nets, boat strikes, the risk of cold-stunning, and the ingestion of ocean plastic (National Research Council 2010; Polovina et al. 2004; The Biology of Sea Turtles 1997; Oros 2005).

The primary threats to adult olive ridleys in the water are incidental take in fisheries and boat collisions, while the primary threat to olive ridleys on nesting beaches is the illegal harvest of turtles and eggs on Mexican and Central American nesting beaches (NMFS and USFWS 1998). In this study area, the two beaches are located close to Corcovado National Park, meaning the predatory animals that extend their hunting range onto the beach habitats are diverse and abundant (Korein et al. 2019). The greatest threat of predation comes from wild species such as coatis, raccoons, crabs, and birds, as well as domestic animals such as dogs (NMFS and USFWS 1998; Barquero-Edge 2013; Drake 1996; Heithaus 2013). Though not as significant, erosion from storms and seasonal changes can limit available nesting habitat, hinder the movement of hatchlings, and/or wash away nests (NMFS and USFWS 1998). This tends to be a problem on Playa Piro, where nests that are located close to the river and are in danger of being washed away are relocated to another location on the beach or to the hatchery (M. Garrido, personal communication, March 8, 2018). Lastly, in recent decades, this region of the Osa Peninsula has undergone development due to tourism and the gold mining industry (Korein et al. 2019).

CONSERVATION

In response to historic population declines of sea turtles, nightly nest monitoring and establishment of hatcheries have become a popular management strategy for nesting beaches all over the world (SWOT Scientific Advisory Board 2011), in an effort to increase hatching and emergence success of turtle eggs (Drake 1996; Barquero-Edge 2013). An important component of population recovery is the protection of nesting females, their nests, and hatchlings (NMFS and USFWS 1998). Management strategies to increase hatch and emergence rate of neonate turtles include: reducing or eliminating take of turtles and their eggs by implementing public education efforts and/or increasing law enforcement; relocation of nests to a protected hatchery; minimizing impacts of construction; reducing the effects of artificial lighting on hatchlings; and reducing nest predation by predator control and/or covering of nests with predator excluders (NMFS and USFWS 1998). Since 2013, the dominant conservation strategy used by Osa Conservation has been the relocation of nests to a protected hatchery located on Playa Piro (M. Garrido, personal communication, March 8, 2018). Researchers typically relocate a nest if it is in danger of being eroded or was partially predated but still contains some viable eggs (M. Garrido, personal communication, March 8, 2018).

METHODS

DATA COLLECTION

Osa Conservation has conducted nest monitoring and protection on Playa Piro and Playa Pejeeperro since 2008. I did not participate in any data collection, so the following methods are written as they were relayed to me by Osa Conservation employees who conducted monitoring.

Osa Conservation conducts their monitoring with the same protocol as other organizations in the region (M. Garrido, personal communication, March 8, 2018). I cite additional information about data collection methods from Korein et al. (2019), who conducted similar nightly monitoring on Playa Carate, a beach in close proximity to Playa Piro. While conducting this study Korein et al. worked with Frontier, a research and conservation organization with which Osa Conservation conducted joint monitoring in the early years of this study.

From 2008 to 2018, researchers collected nightly nest data along two beaches on the Osa Peninsula: Playa Piro (2 km) and Playa Pejeperro (4.5 km) (Figure 1). Patrols were conducted year-round, though monitoring did not take place every night and tended to peak during the high olive ridley nesting season. Monitoring took place at night or in the early morning. During all patrols, researchers walked the length of each beach in search of adult turtle tracks, nesting females, predated nests, and/or hatched nests. They walked the full length of each beach unless they were prevented from completing the patrol, such as during high river levels at Playa Piro or the presence of other people that threatened their safety.

For the purpose of data collection, each nest was defined as one of five types: in situ (IS) which refers to nests that remained in place after the eggs were laid by the female turtle; relocated to the hatchery (RH) which refers to clutches that were moved to the protected hatchery; relocated on the beach (RB) which refers to clutches that were moved to a different area on the beach; predated (PO) which refers to nests that were completely or partially predated by a human, wild or domestic animal; and wild nests (WN) which refers to a nest that was found after hatchlings emerged from the nest cavity at the end of incubation.

Each beach was divided into 100 m sectors and three zones in relation to distance from the water (Zone 1 = closest to water, Zone 2 = in the middle, Zone 3 = vegetation). When a nesting crawl was encountered, the following data were collected: the date/time, section, zone, crawl type (actual nest or non-nesting event/false crawl), species (as determined by track), distance to vegetation (m), environmental conditions, and triangulation data. A crawl was determined to be a false crawl if no egg chamber was found by carefully inserting a stick into the sand. Nest triangulations were carried out on nests between sectors 10 and 20 on Playa Piro, for close monitoring of individual nests and for performing excavations to investigate hatchling success (Osa Conservation, personal communication, April 4, 2019). GPS coordinates are not useful to mark nests in this area because the margin of error (~ 6m) is too large to find a small egg chamber (Osa Conservation, personal communication, April 4, 2019). To triangulate a nest, three plastic tapes (marked with nest number, date, and either a C (Central), W (West), or E (East)) were attached to three sturdy trees. The angle and the distance between the nest and the plastic tape

knot as well as the distance between the ground and the knot were measured (Osa Conservation, personal communication, April 4, 2019).

If an adult female was encountered on the beach, it was tagged after it finished nesting and physical data were recorded: carapace length/width, scutes, scales, wounds, and general health. In addition, data were collected on predated nests (percentage of predation and predator type), by looking for animal tracks, signs of digging, or broken egg shells in the proximity of the nest. The percentage of eggs that were predated was estimated visually for each nest (M. Garrido, personal communication, March 8, 2018).

Wild nests were excavated three days after they emerged. During excavation, the following data were recorded: number of live and dead turtles in and out of the nest, number of live and dead pipped turtles, stage of development if dead, and number of empty egg shells. The clutch size was determined by counting egg shells, unhatched eggs, and alive/dead hatching neonates that were remaining inside the nest. If predation occurred during the incubation of the wild nests, the predated eggs were counted (Osa Conservation, personal communication, April 4, 2019).

Often, nests on Playa Piro that were threatened by erosion from the nearby river or were partially predated were relocated to a section of Playa Piro that was maintained as a protected hatchery. The hatchery was located on a different sector of beach each year and was constantly monitored. Data from the hatchery are available for 2015, 2016, and 2017, and include the same emergence data as the in-situ nests, plus nest temperature, the average egg weight, average hatchling weight and carapace measurements, and overall success rate. Clutch size was determined during excavation by counting egg shells, unhatched eggs, and alive/dead hatching neonates that were remaining inside the nest. Since relocated nests were sometimes partially predated, the clutch size may be on average lower than wild nests. When hatchlings were present in the hatchery, the nest code was noted and a total of 20 random hatchlings were measured per nest (Osa Conservation, personal communication, April 4, 2019). A caliper was used to measure curved carapace length and width. To measure weight, a clean Ziploc bag was clipped onto a handheld microscale, which was calibrated to 0g. One by one, each hatchling or egg is placed in the bag and its weight is recorded (Osa Conservation, personal communication, April 4, 2019).

DATA ANALYSIS

I obtained the dataset from Osa Conservation, under a standard nondisclosure agreement. The data are stored in Excel spreadsheets, for which there is one for each year from 2008 to 2017, except for 2009, when the data collected were not suitable and thus not provided. I omitted the 2008 and 2010 datasets due to a lack of data compared to the later study years. Thus, my analysis incorporates the data collected from 2011 – 2017, as well hatchery data from 2015-2017. Each

spreadsheet is divided into Playa Piro and Playa Pejeperro, and contains the same information for each study area, such as survey effort, encountered female turtles, nesting events and their outcomes.

Specifically, annual data for each patrol and each nesting event include the following:

- 1) *Effort*: days/time spent monitoring beaches and collecting data
- 2) *Patrols*: beach, sector, zone, researcher, date/time a nesting turtle or crawl was encountered, outcome (nest or false crawl), species (as determined by track), distance to vegetation, environmental conditions, how the nest was treated (left in place or relocated), triangulation data; if an adult female was encountered then if a tag was already present on animal (Y/N), the age of the tag, tag number, species, cranial scales, carapacial scutes, carapace length (cm), carapace width (cm), condition of eyes/nostrils/ears, body fat, epibionts, wounds, distance to vegetation, and distance to current tide
- 3) *Predated/disturbed nests*: date/time a nest was encountered, species, sector, zone, distance to vegetation, the percentage of eggs that were predated, and predator species as determined by track or visual confirmation (human, dog, coati/raccoon, crabs, birds, maggots, erosion, or unknown)
- 4) *Wild nests*: emergence date, excavation date, species, sector, zone, excavation inventory: live out of nest, dead out of nest, live in nest, dead in nest, live pipped in eggshell, dead pipped in eggshell, predated eggs, eggs/shells hatched, unhatched eggs, stage of development of dead turtles or embryos, total eggs, hatch success, and emergence success
 - Hatch success = the percentage of hatchlings that successfully hatched out of the egg (hatched shells + pipped live + pipped dead)
 - Emergence success = the percentage of hatchlings that successfully hatched and successfully emerged from the nest (hatched – (dead inside the nest + dead outside the nest) (note that dead outside the nest does not typically occur (Osa Conservation, personal communication, April 4, 2019))
- 5) *Hatchery*: nest code (denotes placement in hatchery), species, original location where eggs were laid on beach, chamber depth/width, number of eggs, relocation date, estimated due date, actual emergence date, excavation date, average egg weight, average hatchling weight, average carapace weight, average carapace width, live out of nest, dead out of nest, live in nest, dead in nest, live pipped in eggshell, dead pipped in eggshell, predated eggs, eggs/shells hatched, unhatched eggs, stage of development, total eggs, hatch success, and emergence success

When an in situ, predated, or wild nest was encountered on the beach, the record was entered into the corresponding spreadsheet to keep separate records of each nest type. Sometimes data

were entered into the incorrect spreadsheet, so during data cleaning I removed any nest type coded as predated, wild nest, or hatchery in the patrols dataset (which should only have false crawl and in situ nests) and transferred it to the correct dataset (predated, wild nest, or hatchery). To account for possible double-counting of nests, I cross-checked the date and time of entry. If there was an existing entry in the correct dataset that matched the date and time in the patrols dataset, I did not add it to the correct corresponding dataset.

Once I sorted all data into the correct spreadsheets, I conducted all subsequent data cleaning and analysis in R Statistical Software. Specifically, I removed columns irrelevant for analysis; removed completely empty rows of data; renamed the columns; revalued blank cells as NA; revalued NA as Unknown (“UKN”) in some situations (such as species or nest type, to preserve the total nest count); standardized the date and time formats; added a column for year and beach; and standardized data into consistent formats. For example, I standardized the species to 5 different codes: CM (green turtle), LO (olive ridley), EI (hawksbill), DC (leatherback), and UKN (unknown). Earlier datasets recorded species as either A or S, which corresponded to either an asymmetrical or symmetrical track, respectively. Olive ridleys leave asymmetrical tracks and green turtles leave symmetrical tracks. As these are the two most abundant nesters in the region, I recoded species recorded as A and S to LO and CM, respectively. I standardized the nest type to 7 different types: IS (in situ), FC (false crawl), PO (predated), WN (wild nest), RB (relocated on the beach), RH (relocated to the hatchery), and UKN (unknown). In most cases, data was considered unknown or NA if the cell was blank or didn’t make sense (e.g., entered in the wrong column).

Aside from analyses involving the total nest count, I conducted analyses for only green turtles and olive ridleys, as there were few data on hawksbills and leatherbacks. Unless otherwise specified, all analyses are based on numbers of actual nests, meaning that false crawls were removed from the dataset.

Since the majority of data is in the form of counts of nests, I conducted chi-square tests to compare data. For continuous data, I conducted nonparametric Wilcoxon tests and Kruskal-Wallis tests. I used nonparametric tests because the sample sizes between olive ridley nests and green turtle nests were unequal.

RESULTS

EFFORT

Overall, the patrol effort both between beaches and across years was unequal, mostly due to the size difference between Playa Piro (2 km) and Playa Pejeperro (4.5 km). Because Playa Piro is less than half the size of Playa Pejeperro, often two patrols were conducted per night on Playa Piro, typically one at night and one in the morning. Using the number of unique nights spent patrolling

as an indicator of effort, Playa Piro had the greatest effort in 2016 with 328 nights and the lowest effort in 2013 with 96 nights, while Playa Pejeperro had the greatest effort in 2014 with 216 nights and the lowest effort in 2017 with 51 nights (Figure 2). However, when the time spent patrolling per year is used as a measure of effort, Playa Pejeperro was generally higher (Figure 2). This is because time spent patrolling is a reflection of the number of nests that were encountered, and because Pejeperro is twice the size of Playa Piro, patrols took longer and more nests were often encountered.

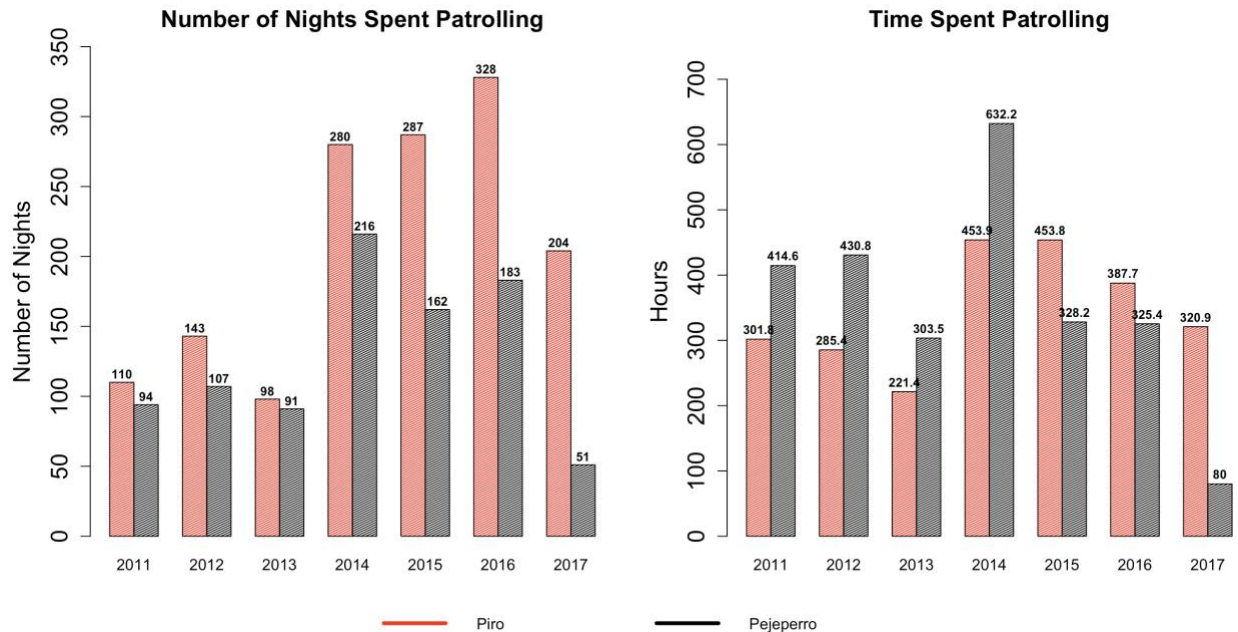


Figure 2: Effort as depicted by number of nights spent patrolling per year (left) and time in hours spent patrolling per year (right) for both Piro and Playa Pejeperro.

Though effort was unequally distributed across years, it was generally equally distributed within the year, or seasonally. At Playa Piro, where patrols were conducted during each month of every year, both the number of patrols and total patrol time peaked during the high nesting season for olive ridleys (July – November) (Figures 3 and 4).

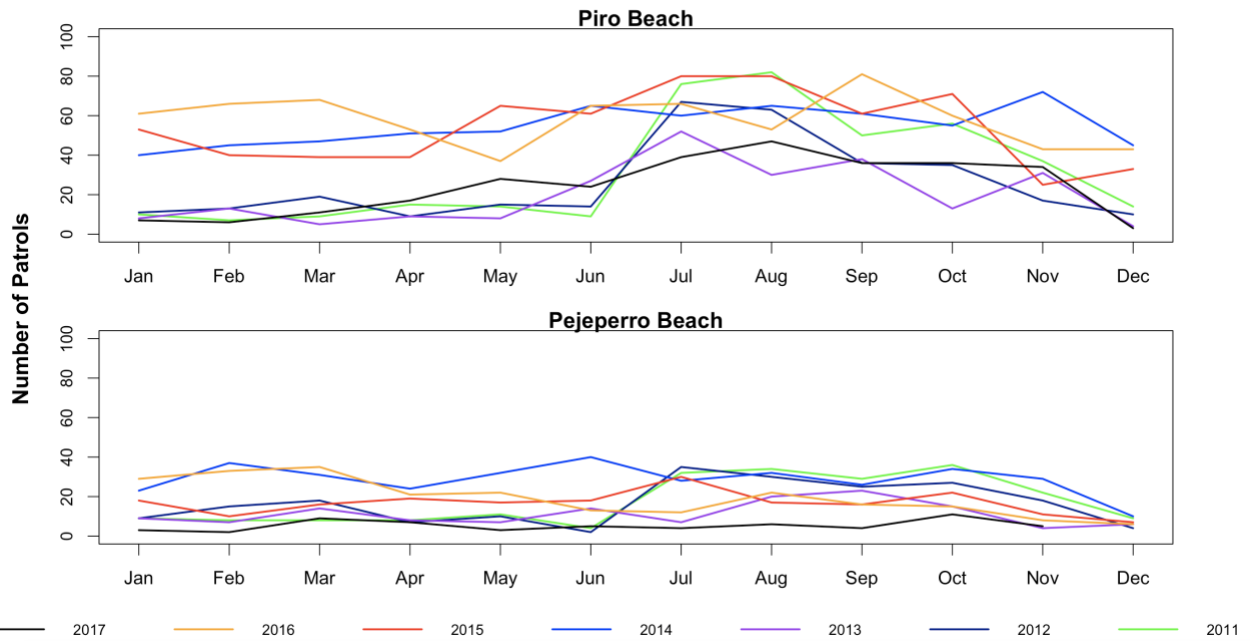


Figure 3: Effort as measured by the number of patrols per month across the study period on Playa Piro (top) and Playa Pejeperro (bottom).

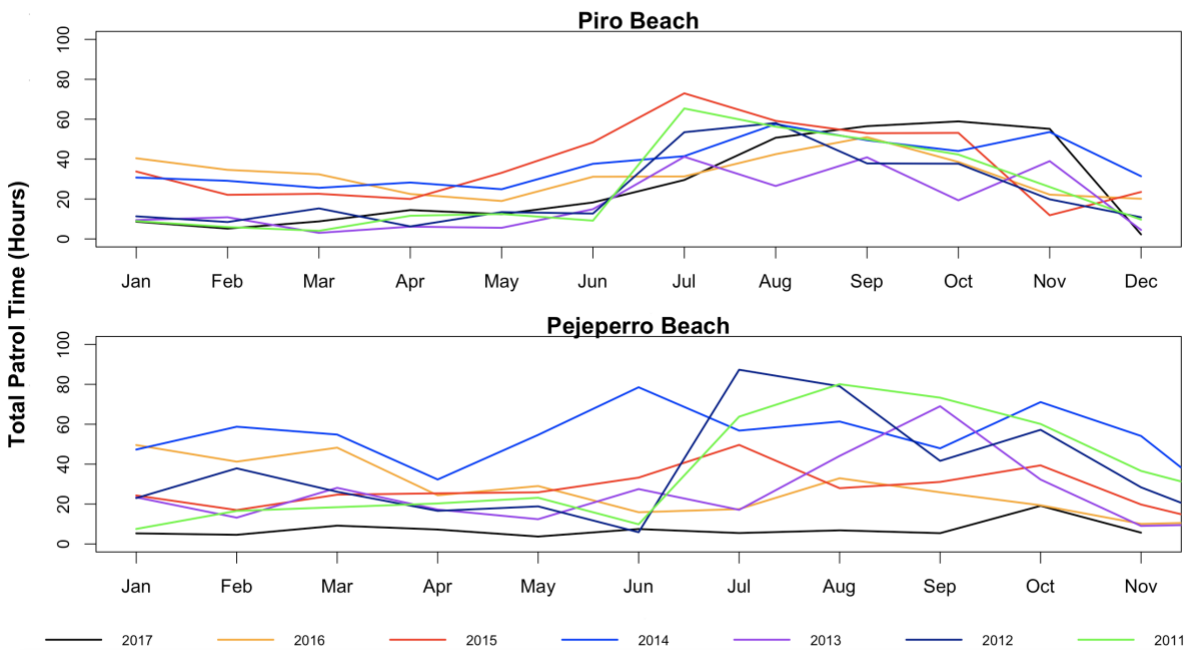


Figure 4: Effort as measured by the total patrol time (hours) per month across the study period on Playa Piro (top) and Playa Pejeperro (bottom).

For the purposes of this report, I did not interpolate the number of nests for nights that lacked patrols. Doing so would have relied on several assumptions and would likely result in the loss of precision, so I conducted the following analyses on the raw number of nests.

NEST CHARACTERISTICS

Over the course of the study period there were 19,113 nesting events on Playa Piro and Playa Pejeperro, with 7,535 occurring on Piro and 11,578 occurring on Pejeperro. After removing false crawls, Piro had 5,777 nests while Pejeperro had 8,367. The type of nest is as follows: at Playa Piro, 49% of nests were in situ, 28% were predated, 13% were relocated to the hatchery, 8% were emerged wild nests, and < 2% were unknown or relocated on the beach (Figure 5). At Playa Pejeperro, 75% of nests were in situ, 18% were predated, 6% were emerged wild nests, and 2% were unknown (Figure 5).

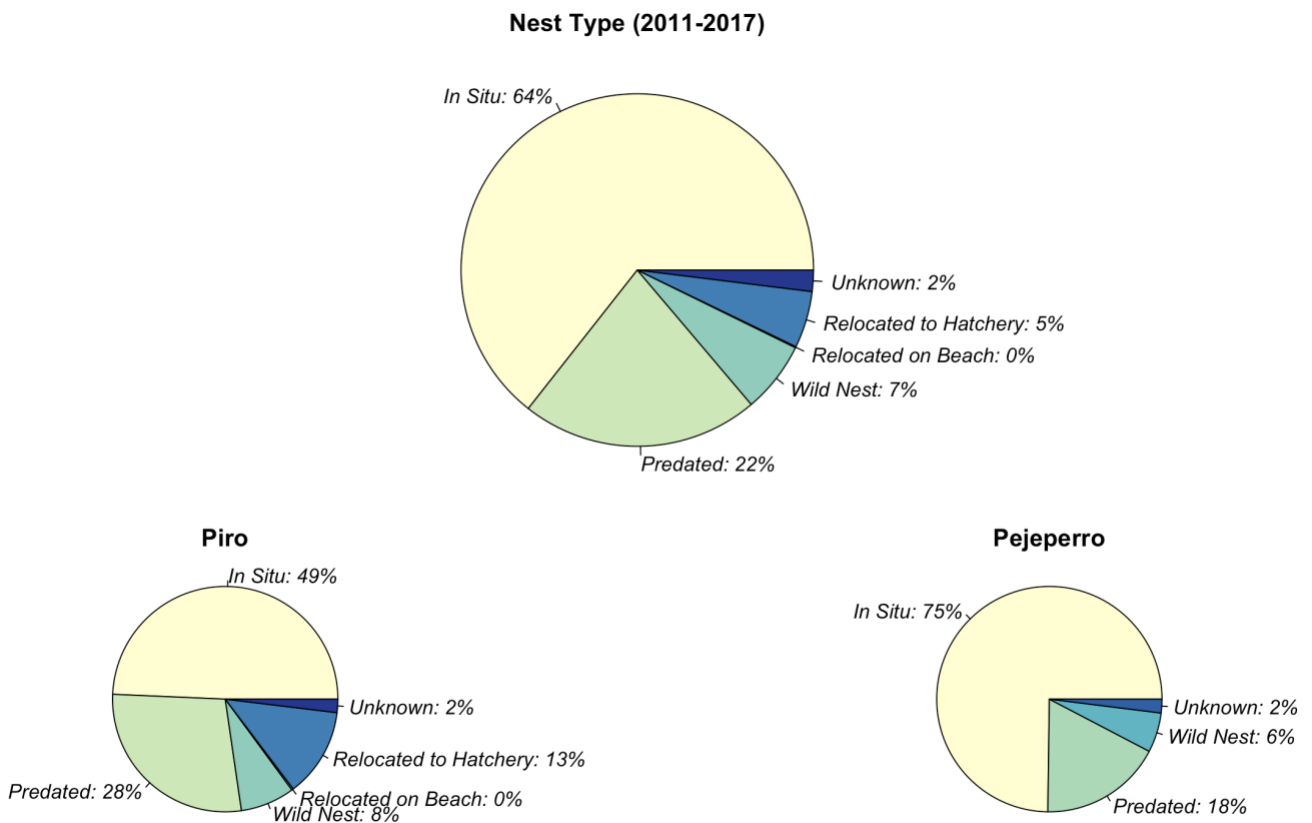


Figure 5: Percentage of each type of nest out of total nest count for Playa Piro (bottom left), Playa Pejeperro (bottom right), and combined (top). The percent of each nest type is averaged across the study period.

On Playa Piro, 77% of nesting events were olive ridley and 15% were green turtles, while on Playa Pejeperro 64% were olive ridley and 28% were green turtles. Hawksbills infrequently nested on Playa Piro and Playa Pejeperro, while leatherbacks infrequently nested on Playa Pejeperro (Figure 6). The greatest number of nesting events occurred in 2014 at both Piro and Pejeperro, and the least number of nesting events occurred in 2011 for Piro and 2017 for Pejeperro (Figure 6). The total nest count per month at each beach indicates that the most active nesting season for olive ridleys is July - November, with the peak typically occurring in September. The most active nesting

season for green turtles is less obvious due to a much lower number of nests, but appears to peak from November - January (Figure 7).

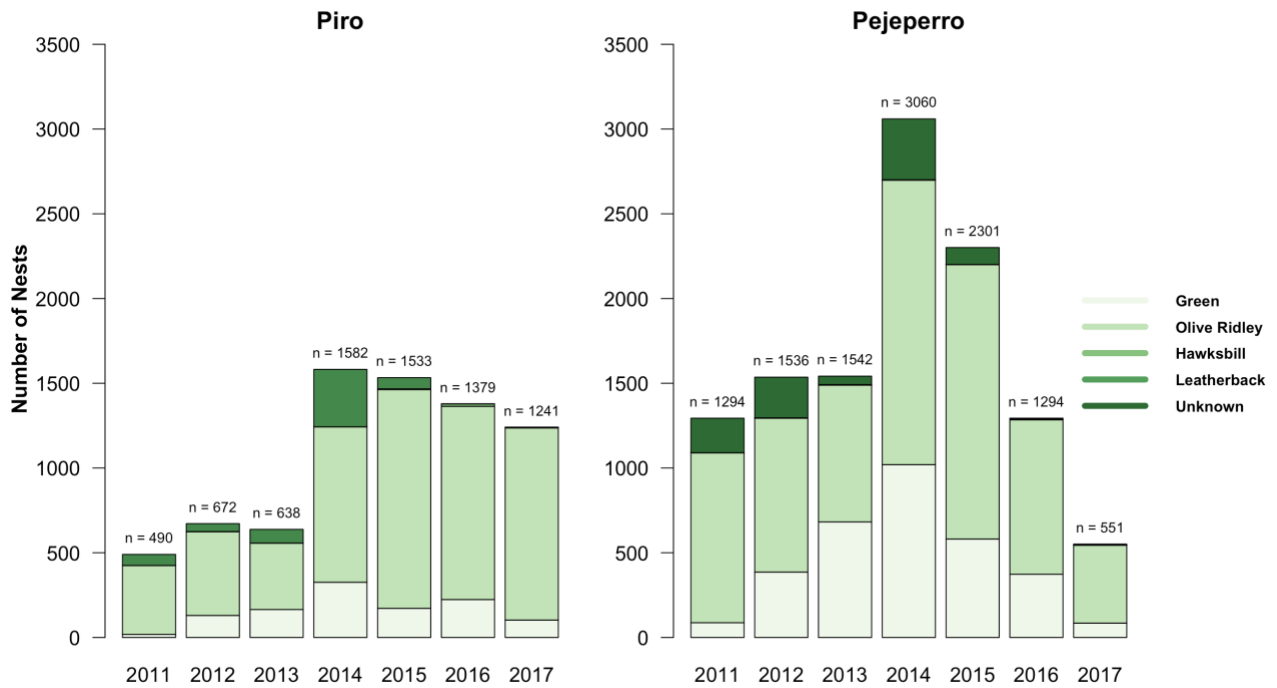


Figure 6: Number of nesting events per species, per year, on Playa Piro (left) and Playa Pejeperro (right). Total count per year is indicated at top of each bar. Nesting events include false crawls.

The distribution of nests across the beaches differed: at Piro, the highest frequency of nests occurred between sectors 8 and 12 (Figure 8); at Pejeperro, nests were more evenly distributed across the sectors, with the highest frequency occurring around sector 10 (Figure 8).

The mean distance to vegetation of olive ridley nests was 6.5m (n = 8717), while the mean distance to vegetation of green turtle nests was 1.5m (n=1839). A nonparametric Wilcoxon test indicated that there is a statistically significant difference in distance to vegetation of nests of green and olive ridley turtles ($p < 0.0001$), in that green turtle nests were less than half as far from the beach vegetation than olive ridley nests (Figure 9). In addition, a chi-square test indicated that nests of olive ridleys and green turtles are not equally distributed among the three nesting zones ($\chi^2 = 717.26$, $df = 2$, $p < 0.0001$), with green turtles preferring zone 3 (vegetation) and olive ridleys preferring zone 2 (middle beach) (Figure 10). Of the 13,130 nests for which distance to vegetation was recorded, 1695 were recorded at the “lagoon”, which is located at Playa Pejeperro, the “river” which is located at Playa Piro, or the “cliff”, which is located at Playa Pejeperro. The majority of these nests (n = 1601) were olive ridley nests that were located at the lagoon (n = 1181). The lagoon at Playa Pejeperro is distributed from sectors 7 to 16 and the river

mouth at Playa Piro changes among sectors 1 through 9 (Osa Conservation, personal communication, April 4, 2019).

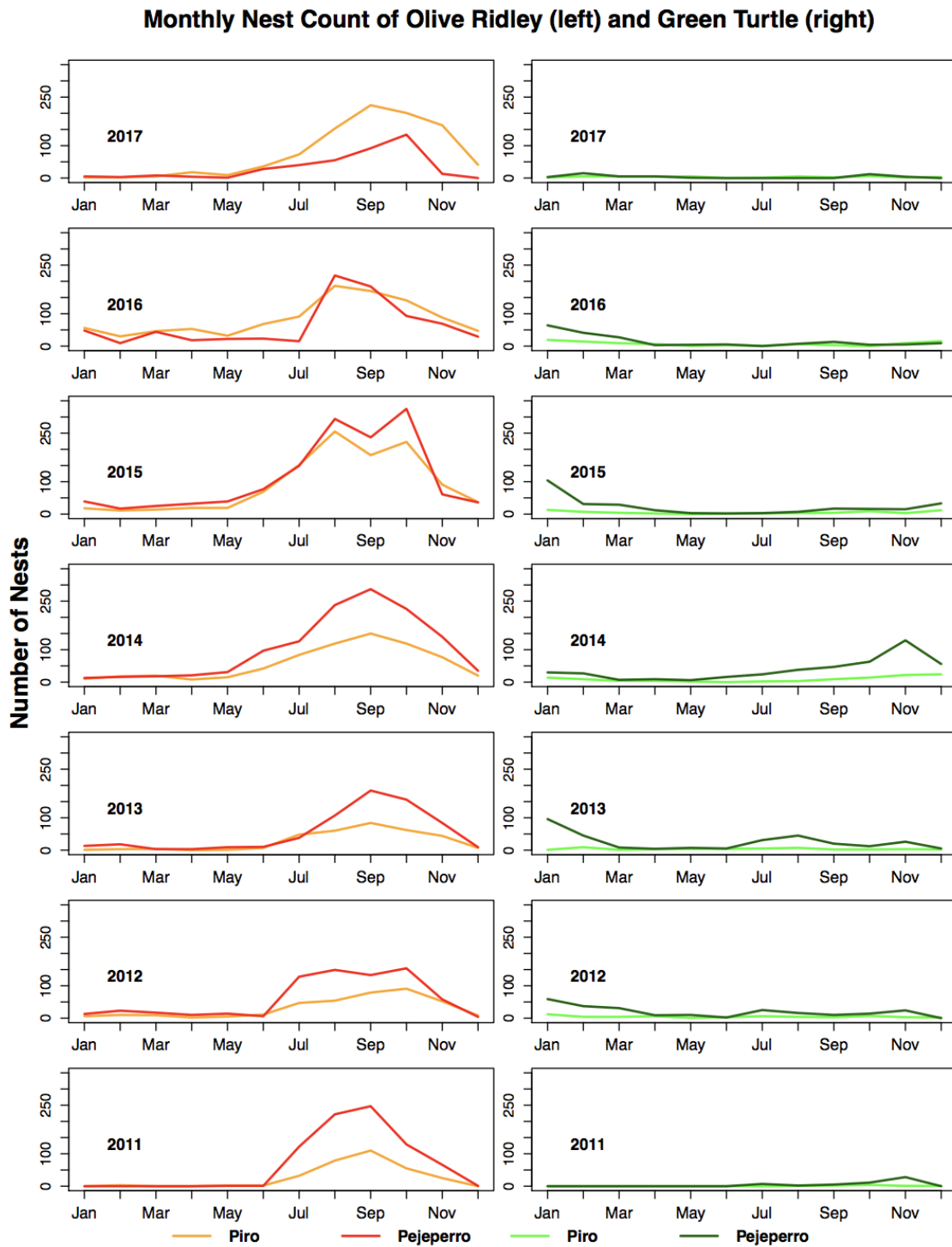


Figure 7: Seasonal nesting trends for the olive ridley (left) and green turtle (right) as depicted by total nest count per month at Playa Piro and Playa Pejeperro.

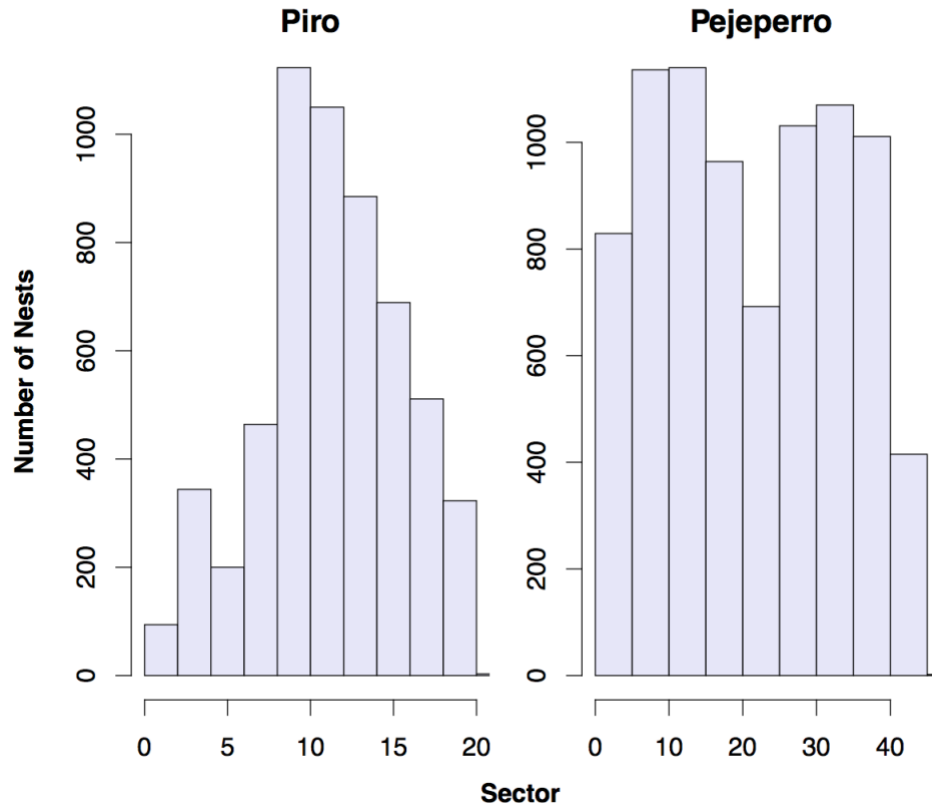


Figure 8: The frequency of nests within each sector of beach for Playa Piro (left) and Playa Pejeperro (right). Each sector is a 0.1 km section of beach. Playa Piro has 20 sectors (2 km) and Playa Pejeperro has 45 sectors (4.5 km).

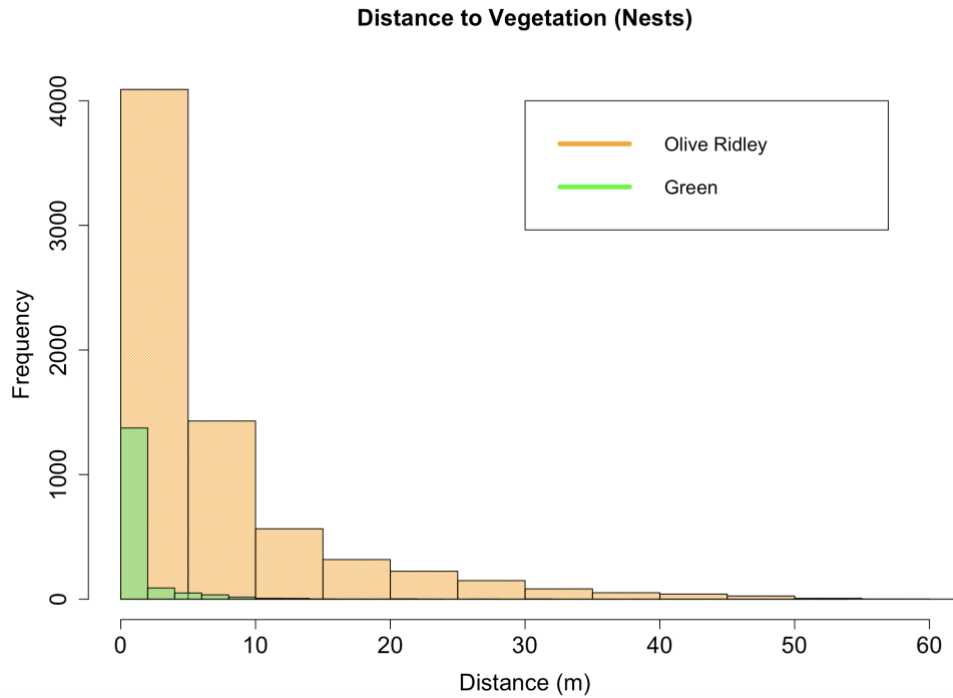


Figure 9: The frequency of olive ridley and green turtle nests in relation to distance from vegetation (m) across the study period.

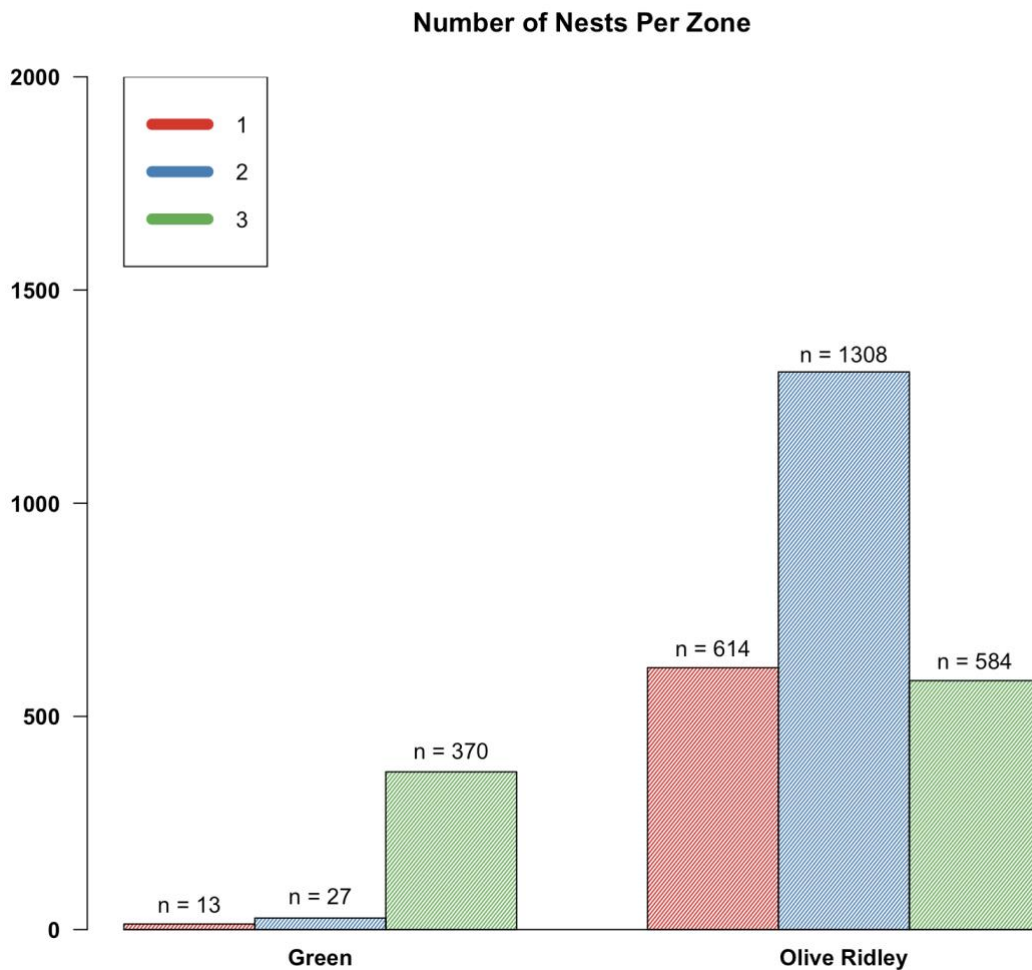


Figure 10: The number of olive ridley and green turtle nests in each beach zone from 2015-2017. Zone data was not collected in earlier years. Zone 1 = closest to water, Zone 2 = in the middle, Zone 3 = vegetation.

PREDATION

Across the study period, 28% of Piro's and 18% of Pejeperro's nests were predated. The percent of predated nests on both beaches has been steadily increasing since 2011 for Playa Piro and 2013 for Playa Pejeperro (Figure 11). For Playa Piro, the predation percentage has risen from 12.3% to 40.7% (Figure 11). Playa Pejeperro experienced high predation in 2012 at 44.4%, but from 2013 to 2017 rose steadily from 2.4% to 48.9% (Figure 11). To explore seasonal trends in predation rates, I calculated the predation percentage per month of humans, dogs, and coati for 2015-2017, as these years represented the most complete set of monthly data. From this analysis, there are no clear patterns of any seasonal trends in predation for any of the three species (Figure 12).

Percentage of Predated Nests

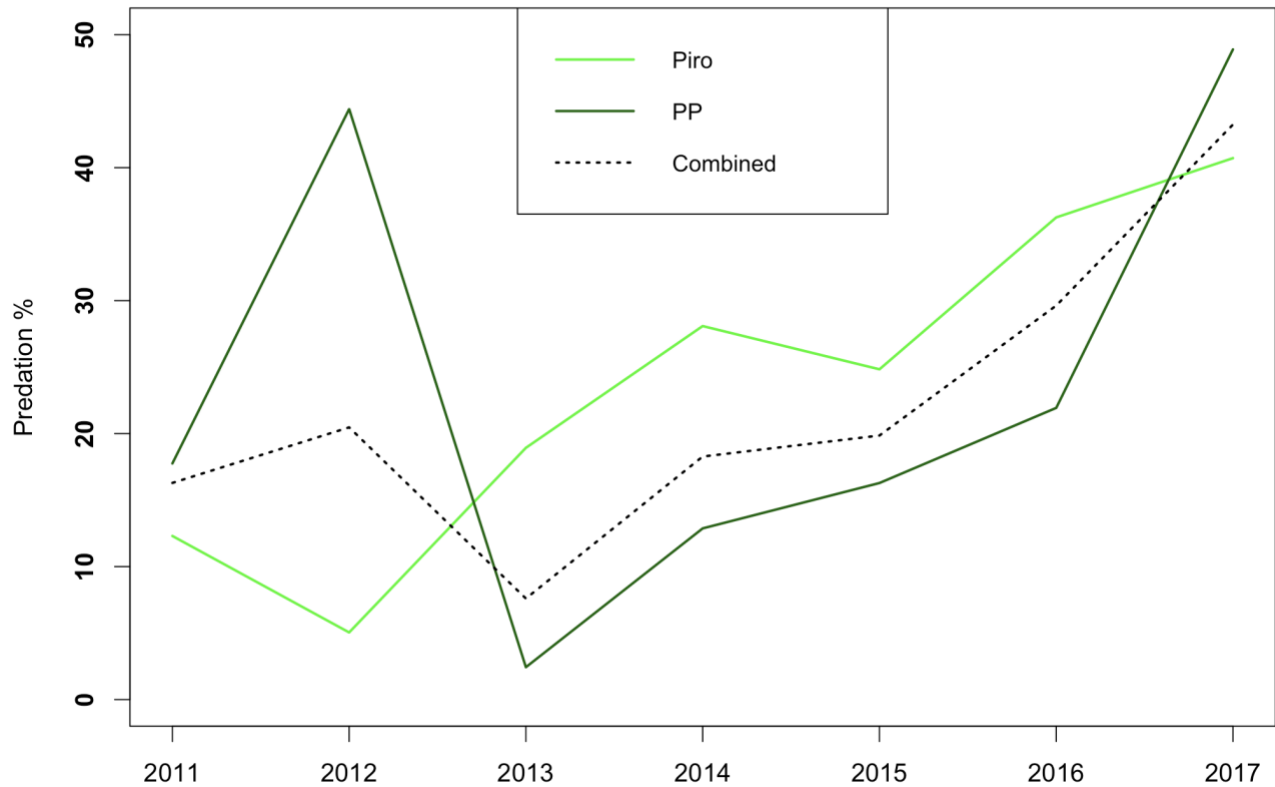


Figure 11: Change in predation rate of nests across study period for Playa Piro, Playa Pejeperro, and their average. The percentage of predated nests was calculated by dividing the number of predated nests by the total number of nests (excluding false crawls) for each year.

During data collection, the predator type was marked as one of 8 categories: humans, dogs, coatis/raccoons, crabs, birds, maggots, erosion, or unknown. To more accurately calculate predation percentage by predator type, I removed eroded nests from the predation dataset and looked at its effect separately. Across the study period, there were 3,087 predated nests. Of these, 2,423 predators were identified (78.5%), while the rest were unknown (21.5%). 2,117 nests were predated by a single predator, while 970 nests were predated by multiple predators. In order to calculate predation percentages out of 100%, I limited my analysis to the nests that were predated by a *single* predator. Of these nests: 173 (8.2%) were predated solely by humans, 429 (20.3%) were predated solely by dogs, 397 (18.8%) were predated solely by coati/raccoon, 145 (6.8%) were predated solely by crabs, 336 (15.9%) were predated solely by birds, 37 (1.7%) were predated solely by maggots, and 600 (28.3%) were unknown (Table 1, Figure 13).

Table 1: Predation percentage of nests by each predator type. The percentage was calculated from the total number of nests predated by a single predator.

beach	humans	dogs	coati/raccoon	crabs	birds	maggots	unknown	total
Piro	114 (10.4%)	122 (11.2%)	301 (27.6%)	73 (6.7%)	168 (15.4%)	30 (2.7%)	283 (25.9%)	1091 (100%)
Pejeperro	59 (5.8%)	307 (29.9%)	96 (9.4%)	72 (7%)	168 (16.4%)	7 (0.7%)	317 (30.9%)	1026 (100%)
Total	173 (8.2%)	429 (20.3%)	397 (18.8%)	145 (6.8%)	336 (15.9%)	37 (1.7%)	600 (28.3%)	2117 (100%)

Monthly Predation Rate at Piro Beach (left) and Pejeperro Beach (right)

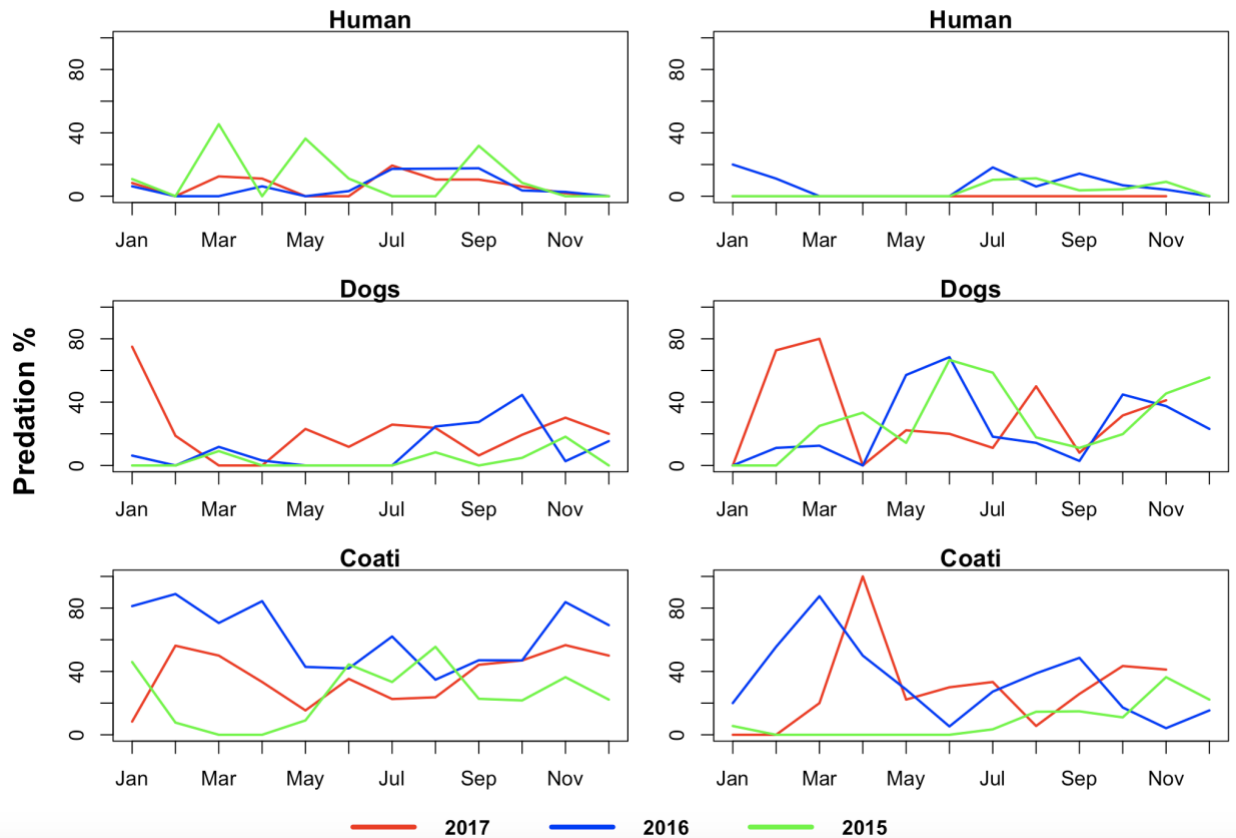


Figure 12: Percent of predated nests by humans, dogs, and coati each month (2015-2017), for Playa Piro (left) and Playa Pejeperro (right).

Of the 3,087 predated nests, 1,617 were at Playa Piro and 1,469 were at Playa Pejeperro. At Playa Piro 1,091 nests were predated by a single predator, while 526 nests were predated by multiple predators. Of the single-predated nests: 114 (10.4%) were predated solely by humans, 122 (11.2%) were predated solely by dogs, 301 (27.6%) were predated solely by coati/raccoon, 73 (6.7%) were predated solely by crabs, 168 (15.4%) were predated solely by birds, 30 (2.7%) were predated solely by maggots, and 283 (25.9%) were unknown (Table 1, Figure 13). At Playa

Pejeperro 1,026 nests were predated by a single predator, while 443 nests were predated by multiple predators. Of the single-predated nests: 59 (5.8%) were predated solely by humans, 307 (29.9%) were predated solely by dogs, 96 (9.4%) were predated solely by coati/raccoon, 72 (7%) were predated solely by crabs, 168 (16.4%) were predated solely by birds, 7 (0.7%) were predated solely by maggots, and 317 (30.9%) were unknown (Table 1, Figure 13).

As both Playa Piro and Playa Pejeperro had approximately the same number of nests predated by a single predator (Piro = 1,091, Pejeperro = 1,026), I conducted chi-square tests to determine if the two beaches experienced significantly different amounts of predation per predator. Overall, the two beaches experienced significantly different amounts of predation from each predator ($\chi^2=217.56$, $df = 6$, $p < 0.0001$). Specifically, Playa Piro experienced more predation by humans ($\chi^2=17.486$, $df = 1$, $p < 0.0001$), coati ($\chi^2=105.86$, $df = 1$, $p < 0.0001$), and maggots ($\chi^2=14.297$, $df = 5$, $p = 0.0001$), while Playa Pejeperro experienced more predation by dogs ($\chi^2=79.779$, $df = 1$, $p < 0.0001$). Both beaches experienced similar rates of predation by crabs ($\chi^2=0.007$, $df = 1$, $p = 0.9338$) and birds ($\chi^2=0$, $df = 5$, $p = 1$). Both beaches also had similar rates of nests impacted by unknown predators ($\chi^2= 1.927$, $df = 1$, $p = 0.1651$).

When considering only nests predated by a single predator, the top three predators at Playa Piro were coati/raccoon (27.6% of nests), birds (15.4%), and dogs (11.2%) (Figure 13). When taking into account all nests with known predators, including those predated by multiple predators, the top three predators remained the same but with higher percentages (particularly birds) (Figure 13). The top three predators at Playa Pejeperro were dogs (29.9%), birds (16.4%), and coati/raccoon (9.4%) (Figure 13). Again, when considering multiple predators, the top three predators remained the same.

I included all predated nests to calculate the predation percentage per predator type for each year of the study. Because some nests experienced predation by multiple predators, the total predation percentage totals were >100%, but the proportion of predation events among years and among predators is still well-represented (Figure 14). Overall, the variation among years in percentage of nests predated by each of these species is highly variable and inconsistent from year to year (Figures 14 and 15).

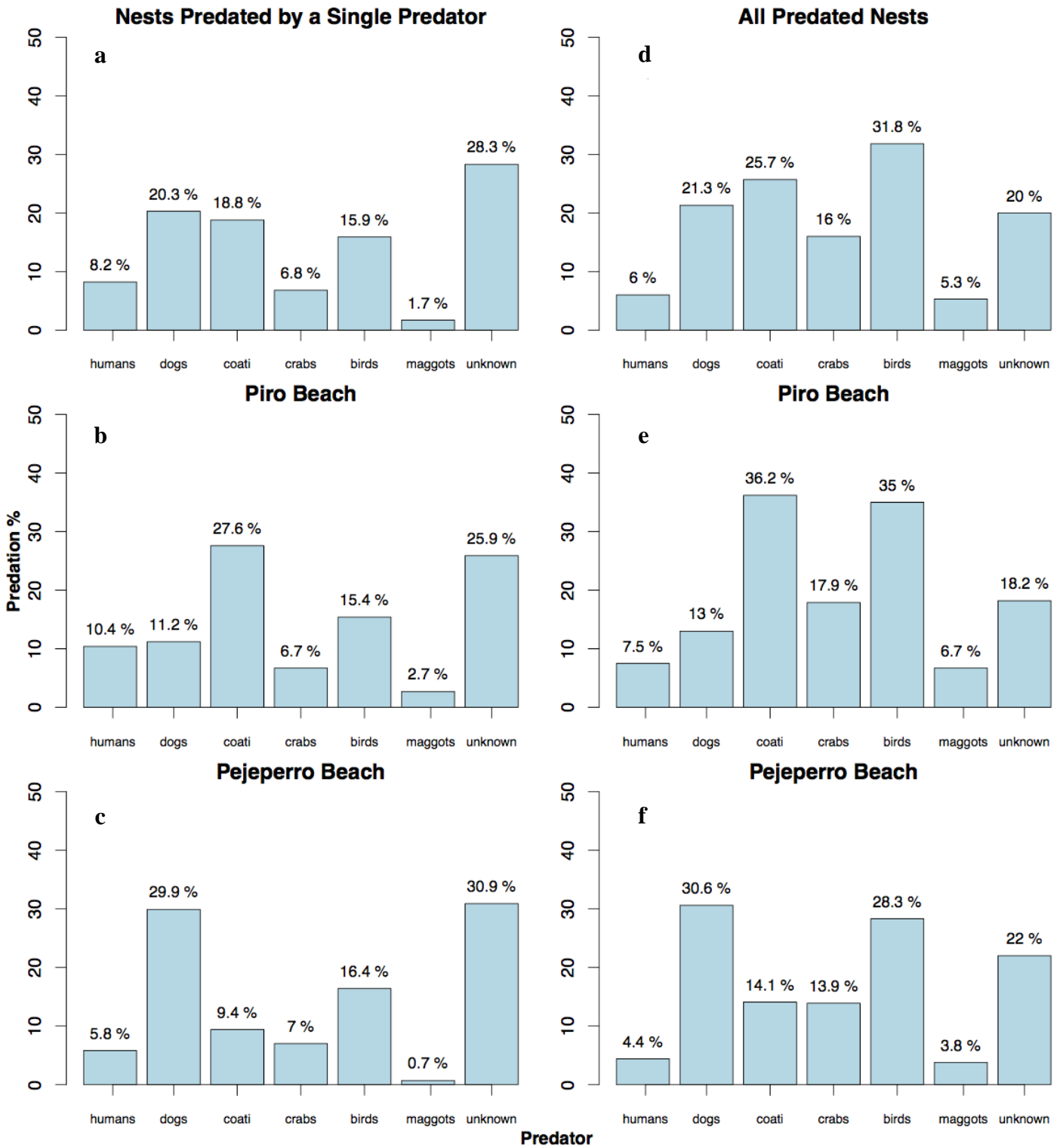


Figure 13: The percentage of nests predated by each predator type, for nests predated by a single predator (left: (a) both beaches (b) Playa Piro (c) Playa Pejeverro) and all nests, including those predated by multiple predators (right: (d) both beaches (e) Playa Piro (f) Playa Pejeverro). These percentages are averaged across the study period.

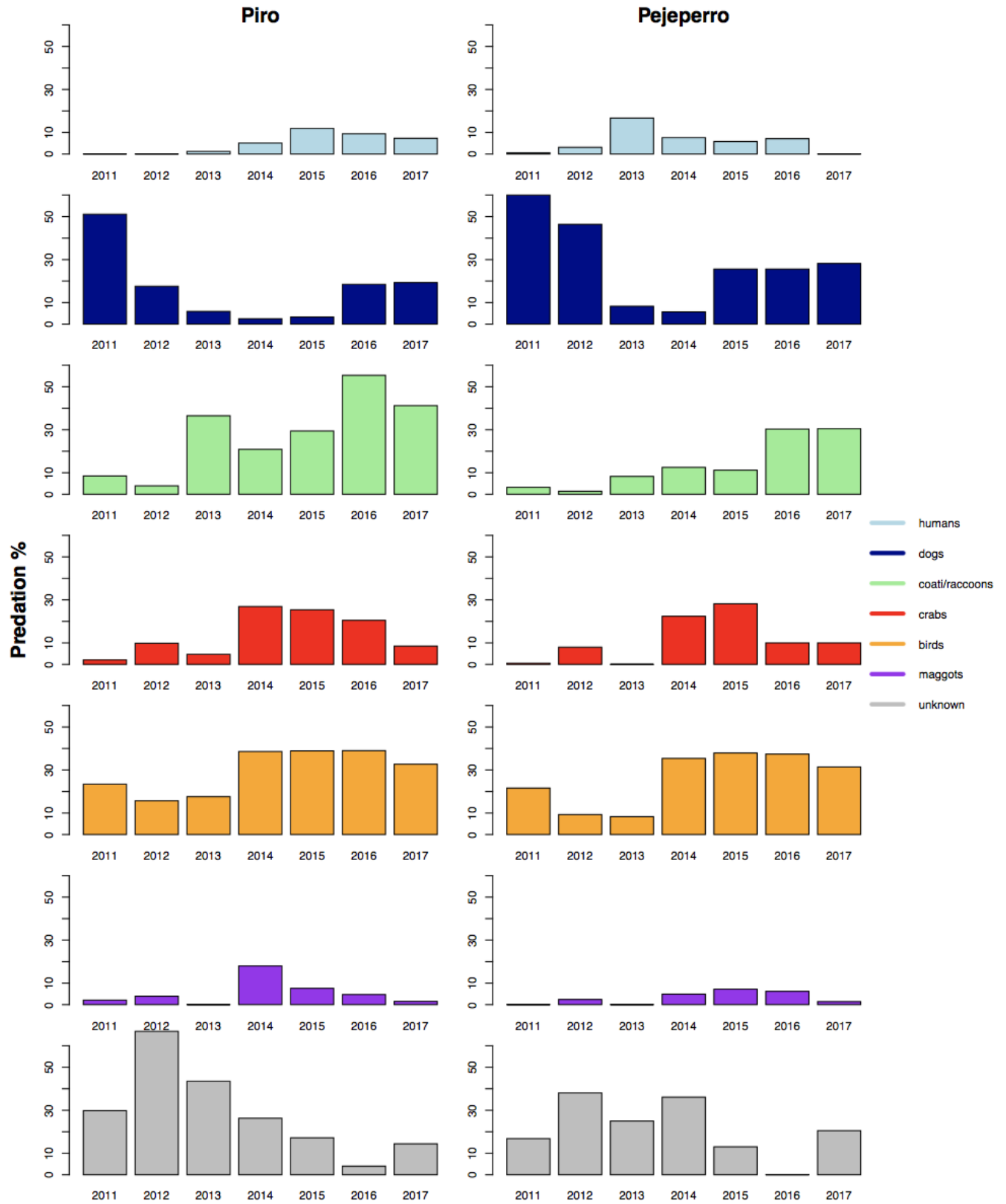


Figure 14: Percentage of all predated nests by each predator type per year for Playa Piro (left) and Playa Pejeperro (right).

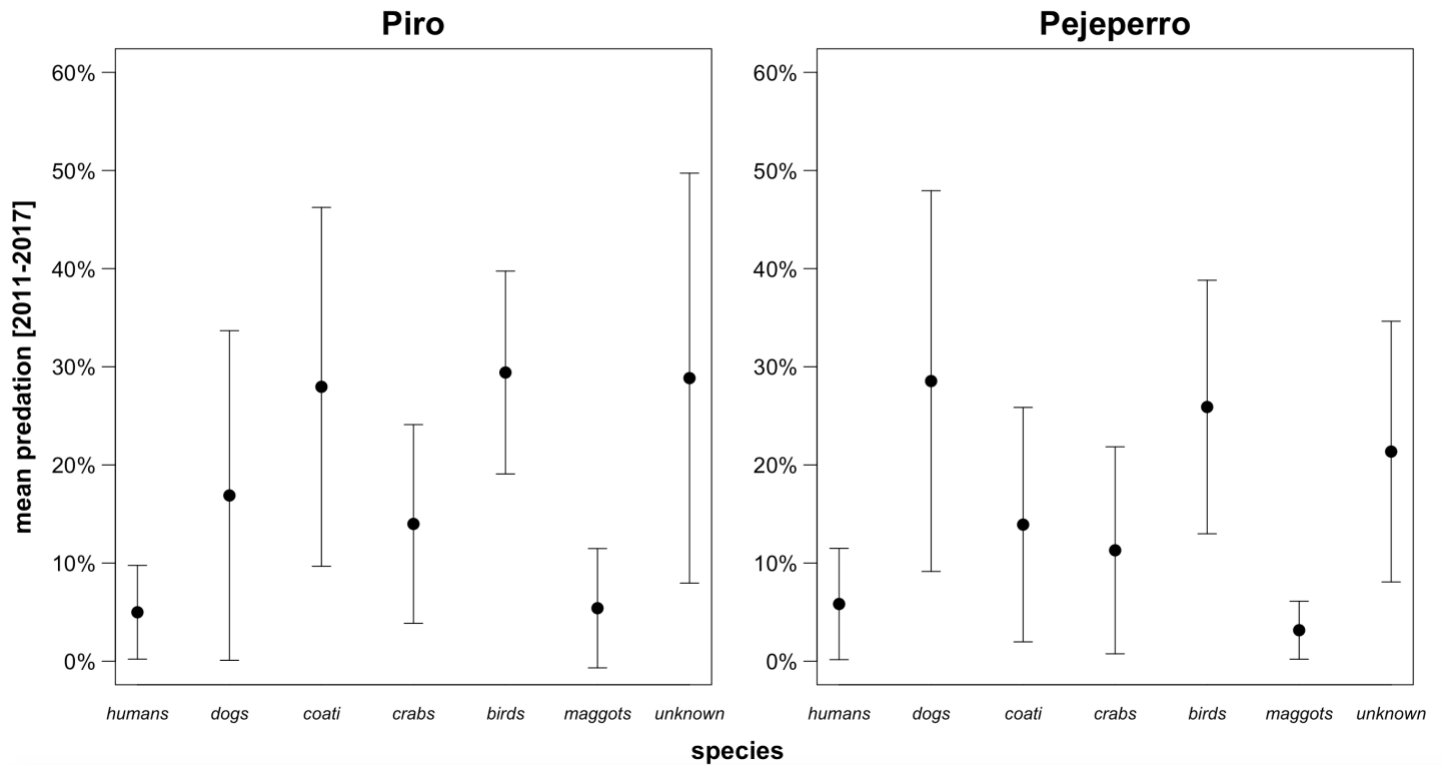


Figure 15: Mean predation percentage and standard deviation of each predator type averaged across all years (2011-2017) for Playa Piro (left) and Playa Pejeperro (right).

Out of 2,327 nests in which the predation percentage was recorded, the mean percent of eggs predated was 69% (SD = 37%), while the median was 100%, meaning that the majority of nests throughout the study period were fully predated with no salvageable eggs.

Table 2: The mean percent of eggs predated in each nest per year.

Year	Mean (%)	SD (%)
2011	100	0
2012	88.6	26.0
2013	66.2	37.7
2014	66.9	35.8
2015	57.0	40.7
2016	67.6	36.6
2017	79.5	31.6

Across the study period, 16 nests were affected by erosion, either primarily or in conjunction with predation. Six of these nests were located at Piro while 10 were located at Pejeperro. Fourteen of these nests were olive ridleys while 2 were unknown species.

LIVE NESTING FEMALES

Across the study period there were 1,398 adult female turtles encountered during patrols on both beaches and recorded. Four hundred ninety-two were recorded on Playa Piro while 924 were recorded on Playa Pejeperro. Two hundred eighty-five (20%) were green turtles, 1,016 (73%) were olive ridleys, 4 were hawksbills (<1%), and 93 (7%) were unknown or the species was not recorded. From 2011 to 2014 the number of turtles encountered each year ranged from 228 to 324. From 2015 to 2017 the number of encountered turtles each year was lower, ranging from 92 to 108 (Figure 16).

Of the encountered turtles, 482 had data recorded about tags. Of these, 33 (7%) had old tags on both their right and left side, 17 (4%) had one old tag and were given a new tag on the other side, and 432 (90%) were untagged and given new tags.

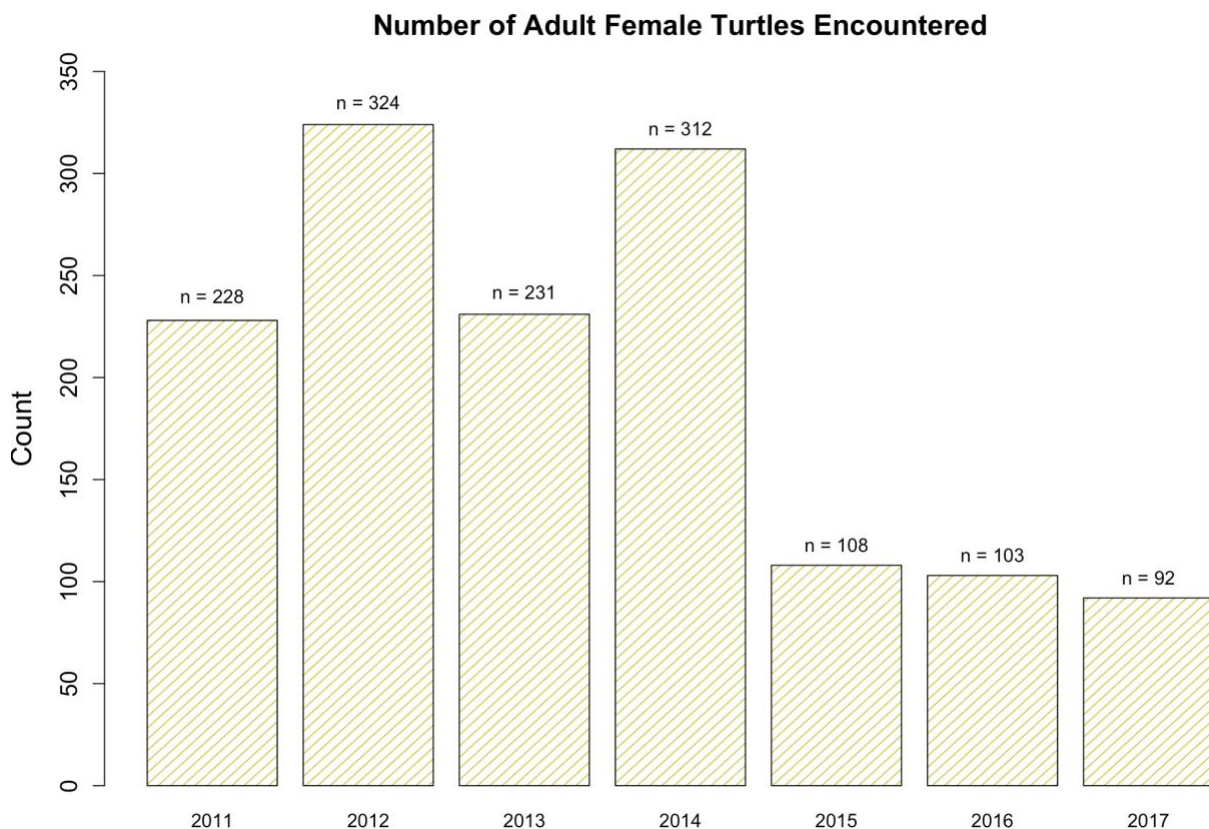


Figure 16: Number of adult nesting females encountered during patrols across both beaches, per year.

Curved Carapace Length

Curved carapace length (CCL) was recorded for 1,215 individuals while curved carapace width (CCW) was recorded for few individuals. I removed all records of CCL below 40 cm from the dataset, since it is highly unlikely that turtles of that size would be nesting and the entry was likely

a mistake. Olive ridleys had a mean CCL of 66.5 cm and green turtles had a mean CCL of 85.9 cm (Table 3). A nonparametric Wilcoxon test indicated that there is a statistically significant difference between the mean CCL of the green turtle and the olive ridley ($p < 0.0001$), in that the mean CCL of the olive ridley is less than that of the green turtle (Figure 17).

Table 3: Minimum, maximum, median, and mean curved carapace length (CCL) for the olive ridley, green turtle, and all recorded turtles.

Curved Carapace Length (cm)				
Species	Min	Max	Median	Mean
Olive Ridley	50	94	66	66.5
Green	51	104	87	85.9
ALL	50	104	67	70.4

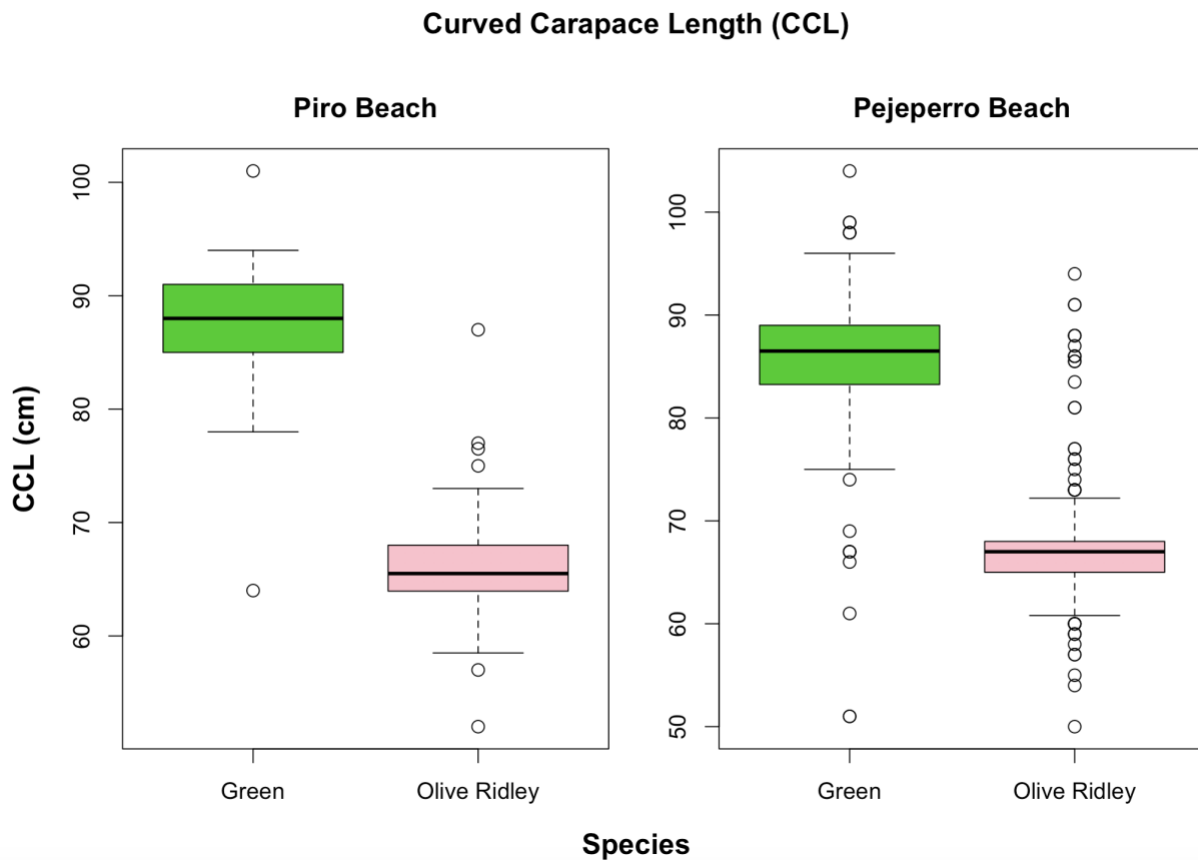


Figure 17: Curved carapace length (CCL) in cm for the olive ridley and green turtle at Playa Piro (left) and Playa Pejeperro (right).

Distance to Tide

The distance to tide was recorded for 889 nesting adults. The mean distance to tide for olive ridleys was 23.5 m and 28.2 m for green turtles (Table 4). A nonparametric Wilcoxon test indicated that there is a statistically significant difference in the recorded distance to tide of adult female nesting green and olive ridley turtles ($p = 0.0002$), in that the distance to tide of the green turtle is greater than that of the olive ridley (Figure 18). There is a statistically significant difference between Playa Piro and Playa Pejeperro in the recorded distance to tide of both adult female green turtles ($p = 0.032$) and adult female olive ridley turtles ($p < 0.0001$), in that the distance to tide at Piro is less than that at Pejeperro (Figure 18). A nonparametric Kruskal- Wallis test indicated that there is a statistically significant difference among years in the recorded distance to tide of adult olive ridley turtles ($p < 0.0001$), but no statistically significant difference among years in the recorded distance to tide of adult green turtles ($p = 0.4644$) (Figure 19).

Table 4: Minimum, maximum, median, and mean distance to tide (m) for the olive ridley and green turtle.

Distance to Tide (m)				
Species	Min	Max	Median	Mean
Olive Ridley (n = 682)	0	84.7	23.1	23.5
Green (n = 167)	0	70.3	28	28.2

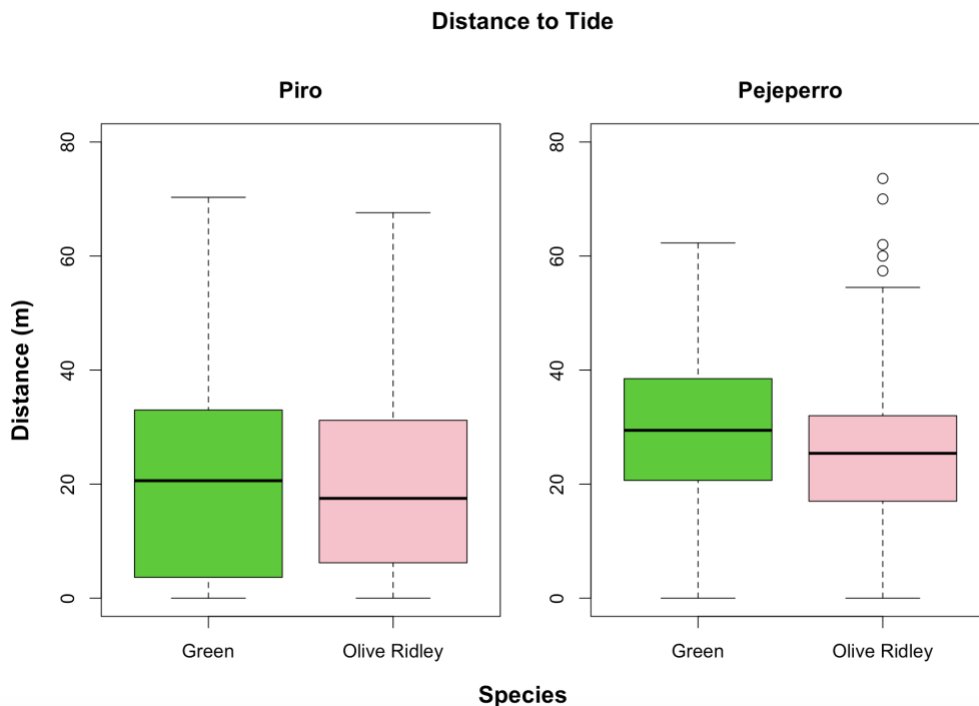


Figure 18: Distance to the tide (m) at which nesting females were recorded on Playa Piro (left) and Playa Pejeperro (right).

Distance to Tide Per Year

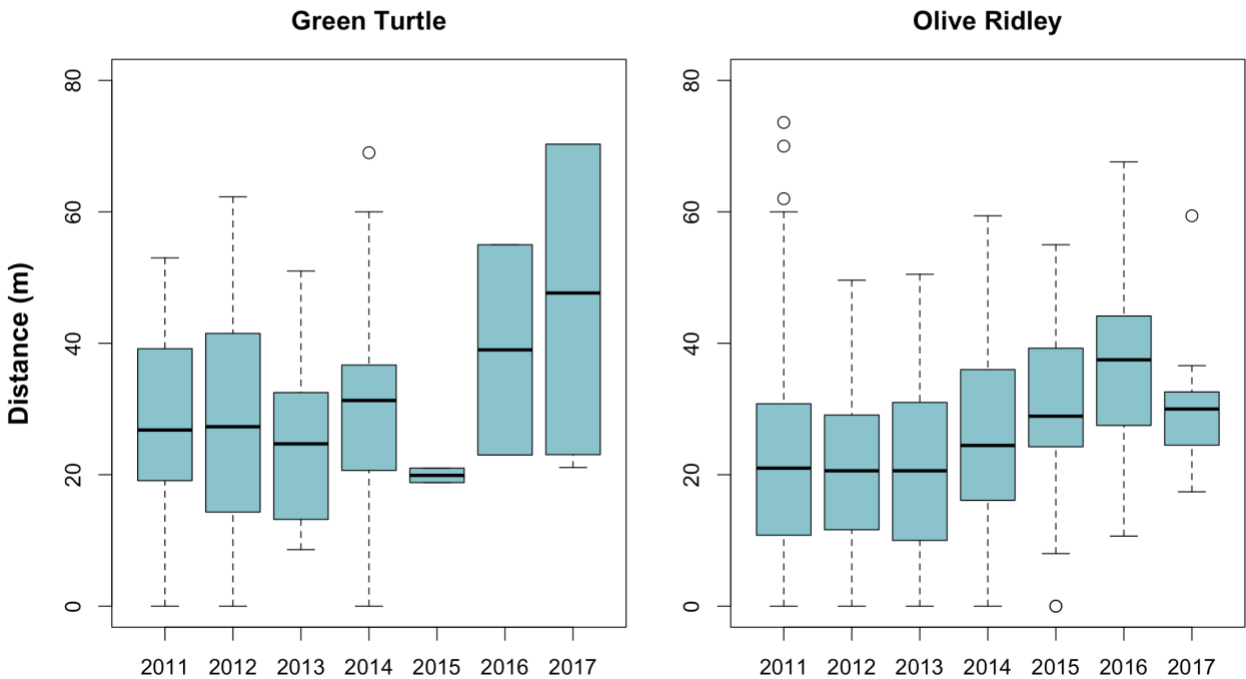


Figure 19: Distance to tide (m) at which nesting females were recorded on both beaches, each year. Bars represent the range of the dataset (excluding outliers), while unfilled circles represent outliers.

Distance to Vegetation

The distance to vegetation was recorded for 1,284 individuals. Of these, 97 were recorded at the river, lagoon, or cliff and were thus removed for this analysis. The mean distance to vegetation was 9 m for olive ridleys and 1.7 m for green turtles (Table 5). A nonparametric Wilcoxon test indicated that there is a statistically significant difference in the recorded distance to vegetation of adult female nesting green and olive ridley turtles ($p < 0.0001$), in that the distance to vegetation of the green turtle is less than that of the olive ridley (Figure 20). There is no statistically significant difference between Playa Piro and Playa Pejeperro in the recorded distance to vegetation of adult female green turtles ($p = 0.6407$) (Figure 20). There is a statistically significant difference between Playa Piro and Playa Pejeperro in the recorded distance to vegetation of adult female olive ridley turtles ($p < 0.0001$), in that the distance at Playa Piro is greater than Playa Pejeperro (Figure 20). A nonparametric Kruskal- Wallis test indicated that there is a statistically significant difference among years in the recorded distance to vegetation of adult green turtles ($p = 0.008$), as well as a statistically significant difference among years in the recorded distance to vegetation of adult olive ridley turtles ($p < 0.0001$) (Figure 21).

Table 5: Minimum, maximum, median, and mean distance to vegetation (m) for the olive ridley and green turtle.

Distance to Vegetation (m)				
Species	Min	Max	Median	Mean
Olive Ridley (n = 927)	0	80	4.9	9
Green (n = 277)	0	42	0	1.7



Figure 20: Distance to the vegetation (m) at which nesting females were recorded at Playa Piro (left) and Playa Pejeperro (right).

Mean Beach Width

I calculated the minimum beach width for each nesting sea turtle as wrack (distance to most recent high tide line) plus toe (distance to primary vegetation). Both measurements were taken for 732 individuals. A Wilcoxon test indicates no statistically significant difference between olive ridleys and green turtles in the mean beach width ($p = 0.172$). However, there is a statistically significant difference between Playa Piro and Playa Pejeperro in the mean beach width ($p < 0.0001$), Playa Piro with a mean of 42.5 m (SD = 18.5) and Playa Pejeperro with a mean of 29.9 m (SD = 12.4). A Kruskal-Wallis test indicates that there is no statistically significant difference in mean beach width among years for neither Piro nor Playa Pejeperro (both beaches: $\chi^2=6$, $df=6$, $p = 0.423$) (Figure 22).

Distance to Vegetation Per Year

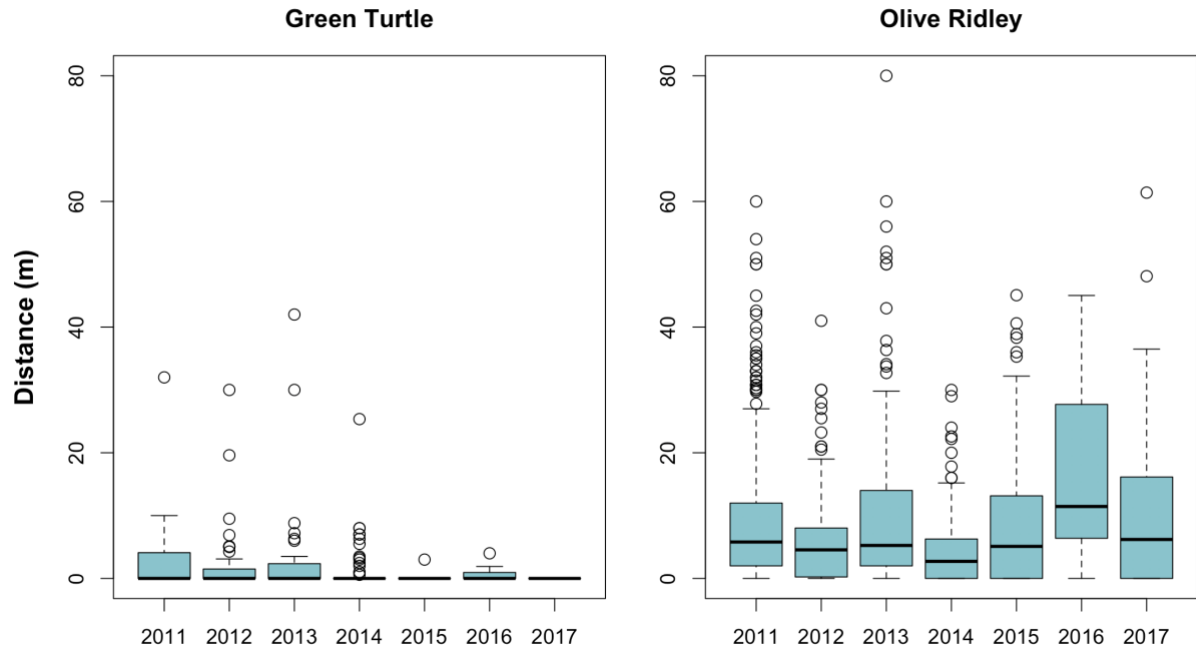


Figure 21: Distance to vegetation (m) at which nesting females were recorded on both beaches, each year. Bars represent the range of the dataset (excluding outliers), while unfilled circles represent outliers.

Mean Beach Width

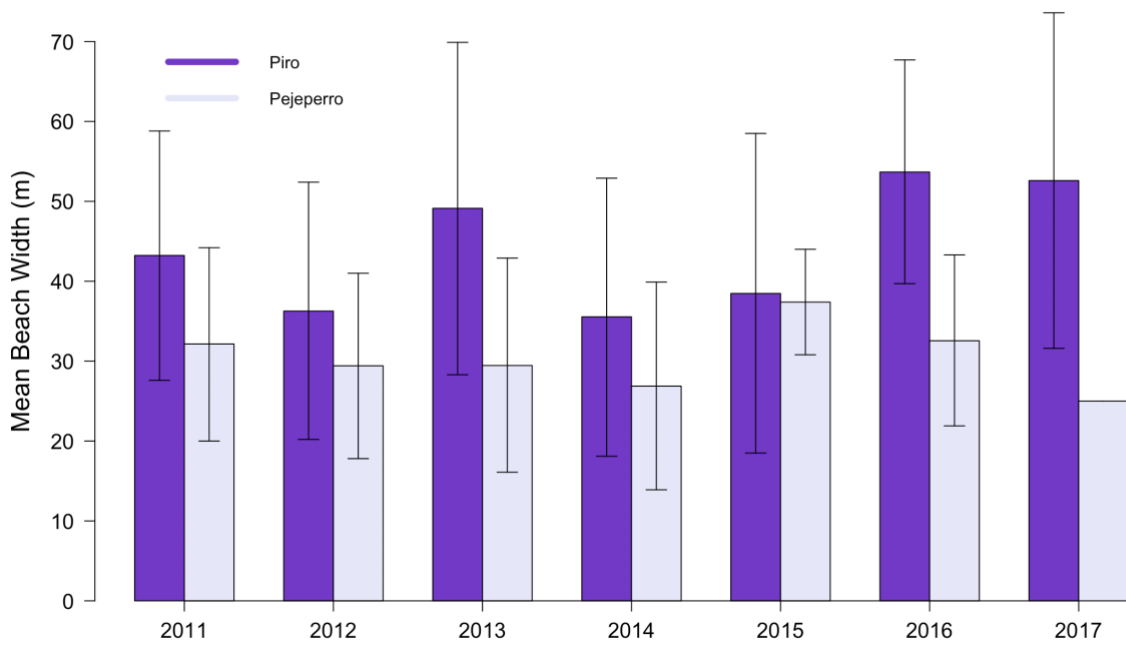


Figure 22: Mean beach width per year, as calculated by averaging the minimum beach width per female (wrack + toe). Error bars represent standard deviation.

WILD NESTS

Hatch success and emergence success were calculated for wild nests during excavation for 2012 – 2017, except for 2014 when the majority of the data was entered into the incorrect spreadsheet and thus did not have any excavation data. Hatch success is defined as the percentage of hatchlings that successfully hatched out of the egg (hatched shells + pipped live + pipped dead), while emergence success is the percentage of hatchlings that successfully hatched and successfully emerged from the nest (hatched – (dead inside the nest + dead outside the nest)). Across the study period the annual mean hatch success ranged from 65.6% - 84.5% and the annual mean emergence success ranged from 62.3% - 78.2%. The lowest success rates occurred in 2013 while the highest occurred in 2016 and 2017 (Figure 23).

I also looked for seasonal trends in success by averaging the hatch and emergence success per month for each year. For months that did not have data on success rates, the trend line automatically interpolated from the previous and following month. There is no year in the study period that shows any seasonal trends in hatch or emergence success, meaning there is variation across the year (Figure 24).

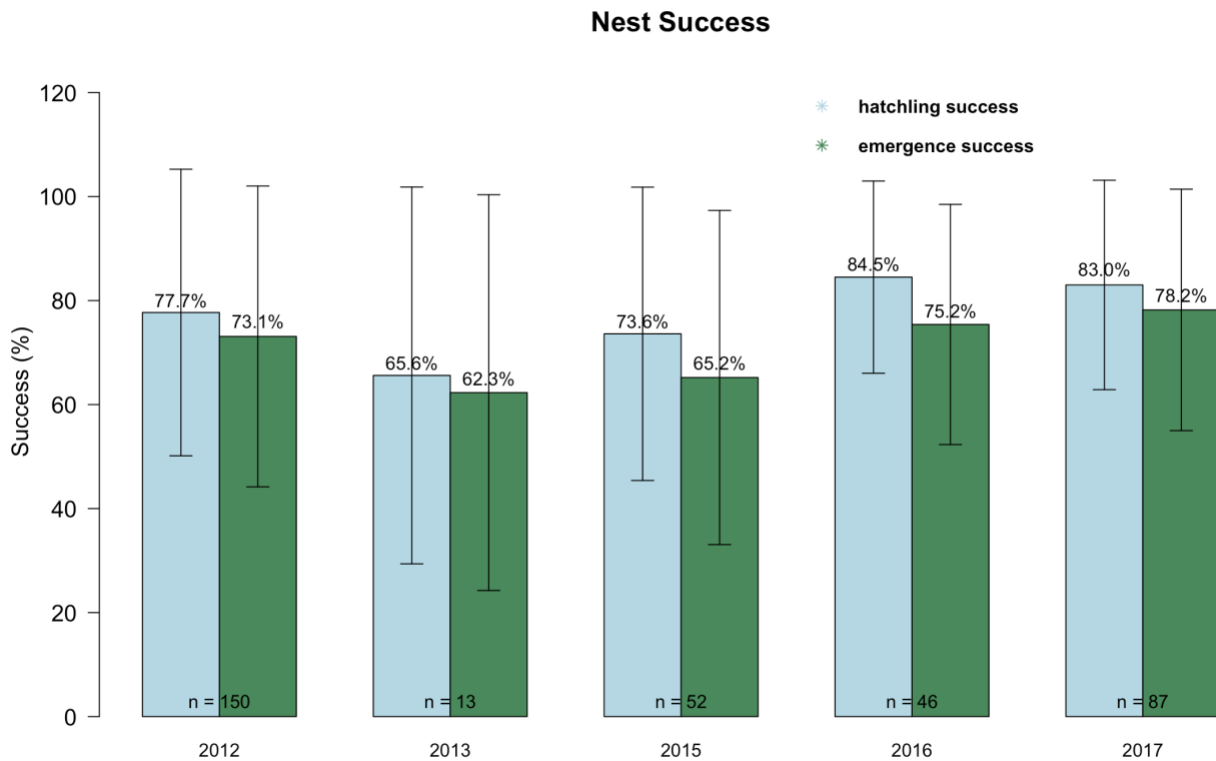


Figure 23: Mean hatch success and emergence success of nests per year for wild (hatched) nests. 2011 did not have any data on nest success, while the majority of the 2014 data was entered into the incorrect spreadsheet and was thus lacking data about success. Error bars represent standard deviation.

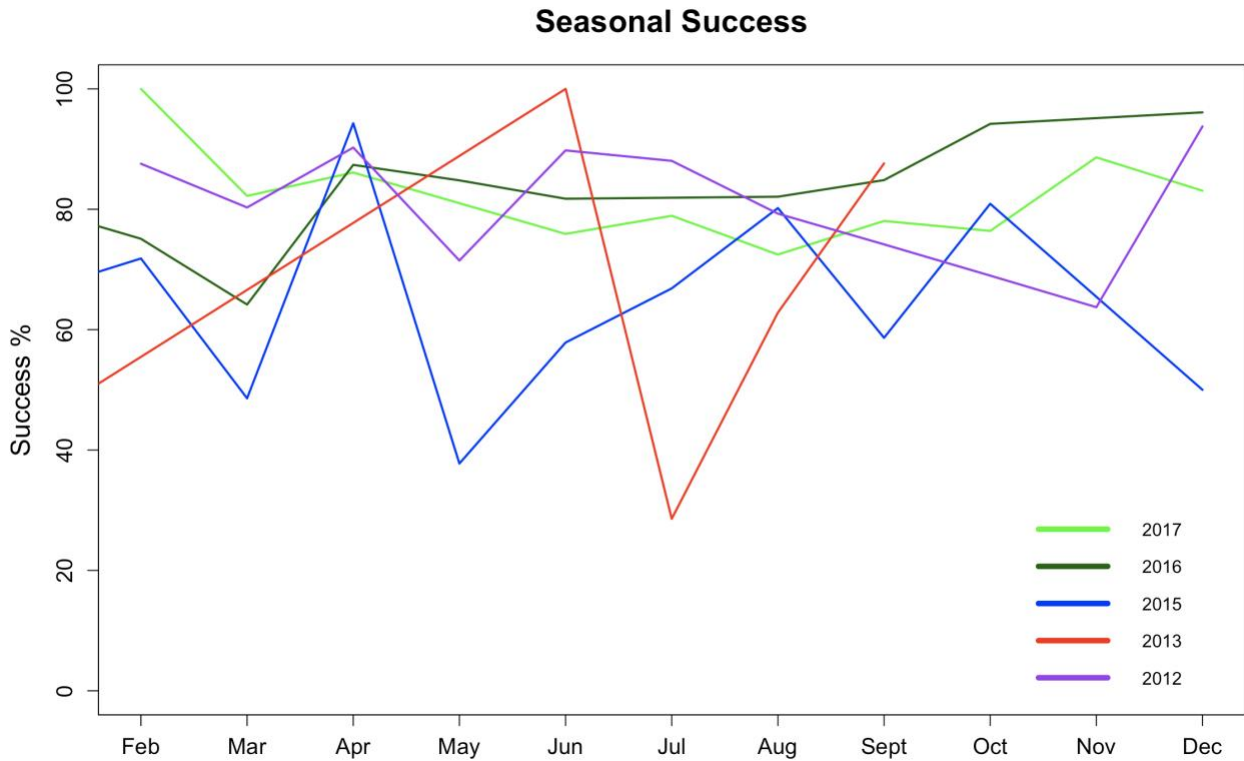


Figure 24: Seasonal trends in success (average of the hatchling and emergence success per month per year). Some months did not have any data, in which case the trend line automatically extrapolated from the previous and following month (except for 2013 which lacked data after September).

Olive ridleys had a mean clutch size of 87.1 eggs (SD = 26.4, n = 343) and green turtles had a mean clutch size of 71.4 eggs (SD = 26.4, n = 68) (Figure 25). A nonparametric Wilcoxon test indicated that there is a statistically significant difference in the mean clutch size of each species ($p < 0.0001$). Eggs counted at “0” were removed from the dataset because it indicated that the nest was not excavated.

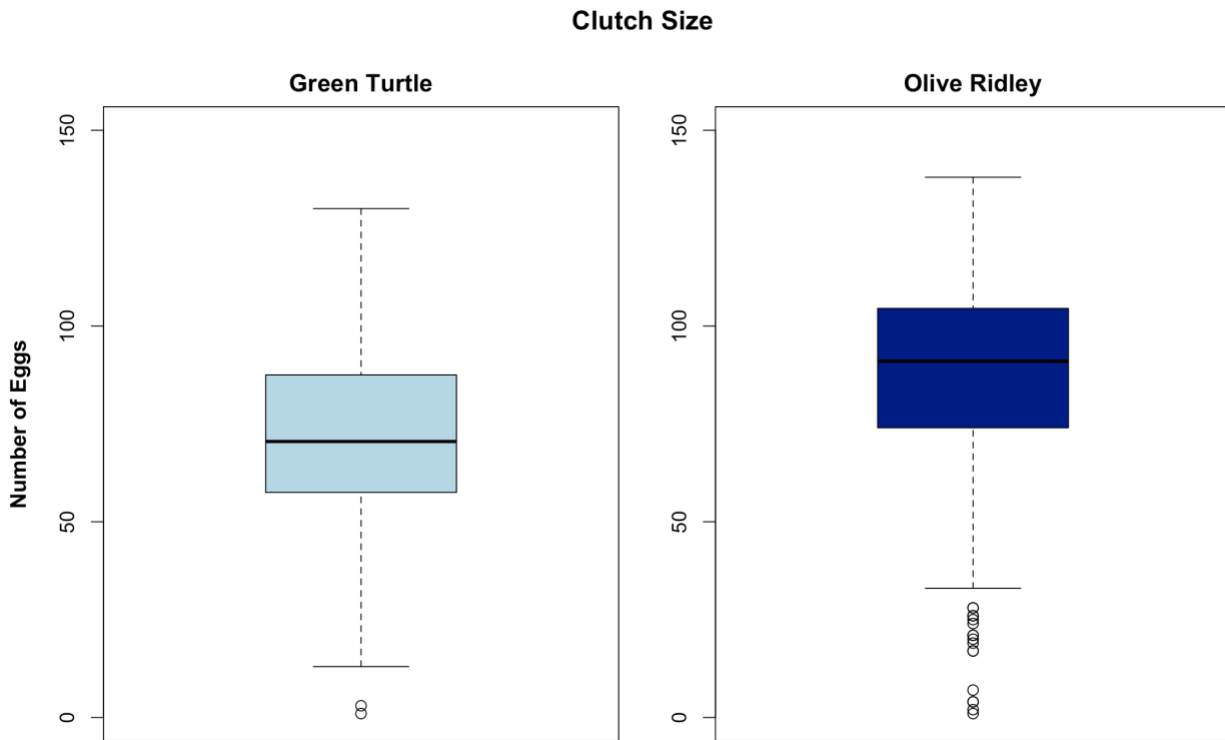


Figure 25: Clutch size distribution of green turtle (left) and olive ridley (right) nests across all years.

HATCHERY

Nests on Playa Piro were relocated to the hatchery if they were threatened by erosion or were partially predated with salvageable eggs. Seven hundred thirty-seven (13%) nests were relocated to the hatchery: 227 in 2015, 206 in 2016, and 304 in 2017. Of the 737 nests, 707 (96%) were olive ridley, 17 (2.3%) were green, 10 were hawksbill (1.4%), and 3 (0.4%) were unknown or not documented. Similar to the wild nests, the hatching and emergence success were calculated during excavation of nests. In addition, the release success was also calculated, which is defined as the number of hatchlings that make it to the ocean in relation to the number of eggs in the nest, both naturally and assisted. Assisted hatchlings didn't emerge from the nest on their own, but instead remained inside the nest and were released by researchers during excavation (Osa Conservation, personal communication, April 4, 2019). The success rates were calculated from nests with a full set of excavation data, which was 181 nests in 2017, 177 nests in 2016, and 204 nests in 2015. One hundred twenty-three nests in 2017, 29 nests in 2016, and 23 nests in 2015 did not have enough data to calculate success. The success rates stayed relatively consistent over the three years, with the hatch success ranging from 85.2% to 86.4%; the emergence success ranging from 80.7% to 81.5%; and the release success ranging from 84.7% to 87.9% (Figure 26).

Hatchery Success

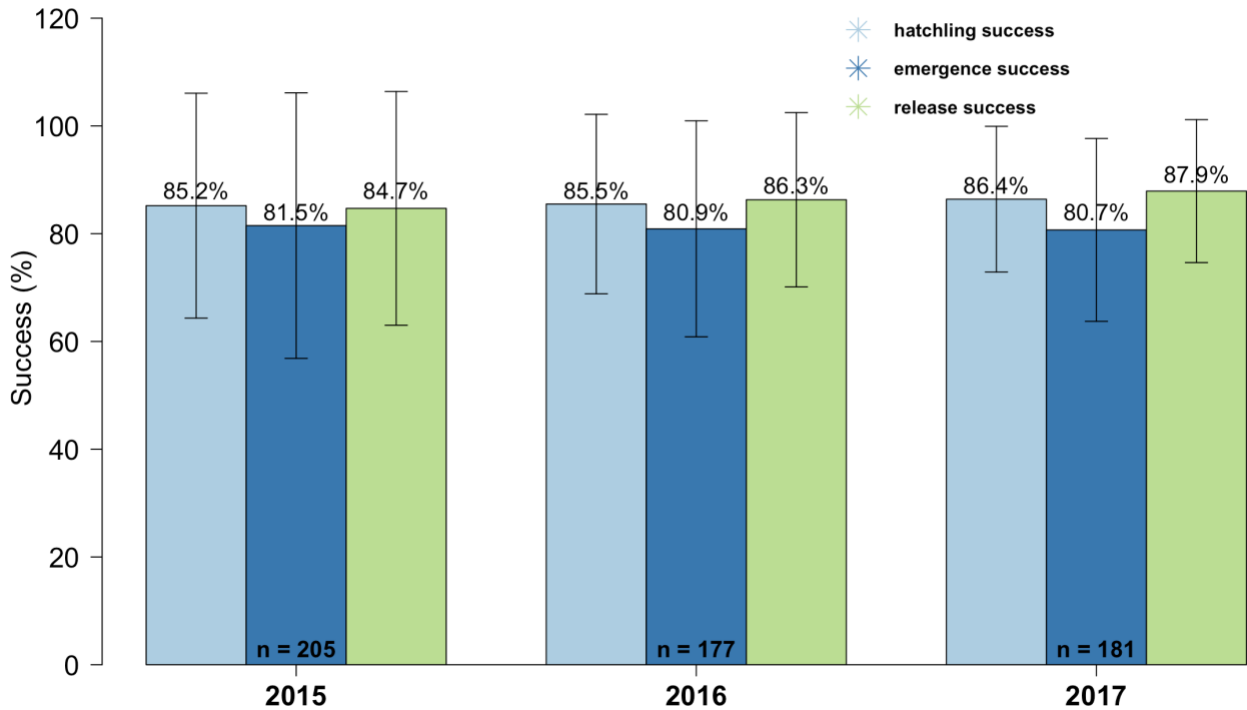


Figure 26: Success rate of hatchery for 2015-2017, as measured by the hatch success, emergence success, and release success averaged across all excavated nests. Error bars represent standard deviation.

The number of hatchlings released was documented for 586 nests. In total, 43,298 hatchlings were released from the hatchery from 2015-2017: 15,569 hatchlings were released in 2015, 13,501 were released in 2016, and 14,228 were released in 2017. This is likely an underestimate because there were an additional 151 nests over the study period that did not have data on the release of hatchlings, but likely produced at least some hatchlings.

I calculated the clutch size distribution for only olive ridley nests, since they made up the majority of the nests in the hatchery. I removed all zeros from the data on the premise that they were mistakes, with the assumption that nests with zero eggs would not be relocated to a hatchery. The mean clutch size was 90.4 eggs (SD = 20.8, n = 613).

Hatchery Clutch Size Distribution

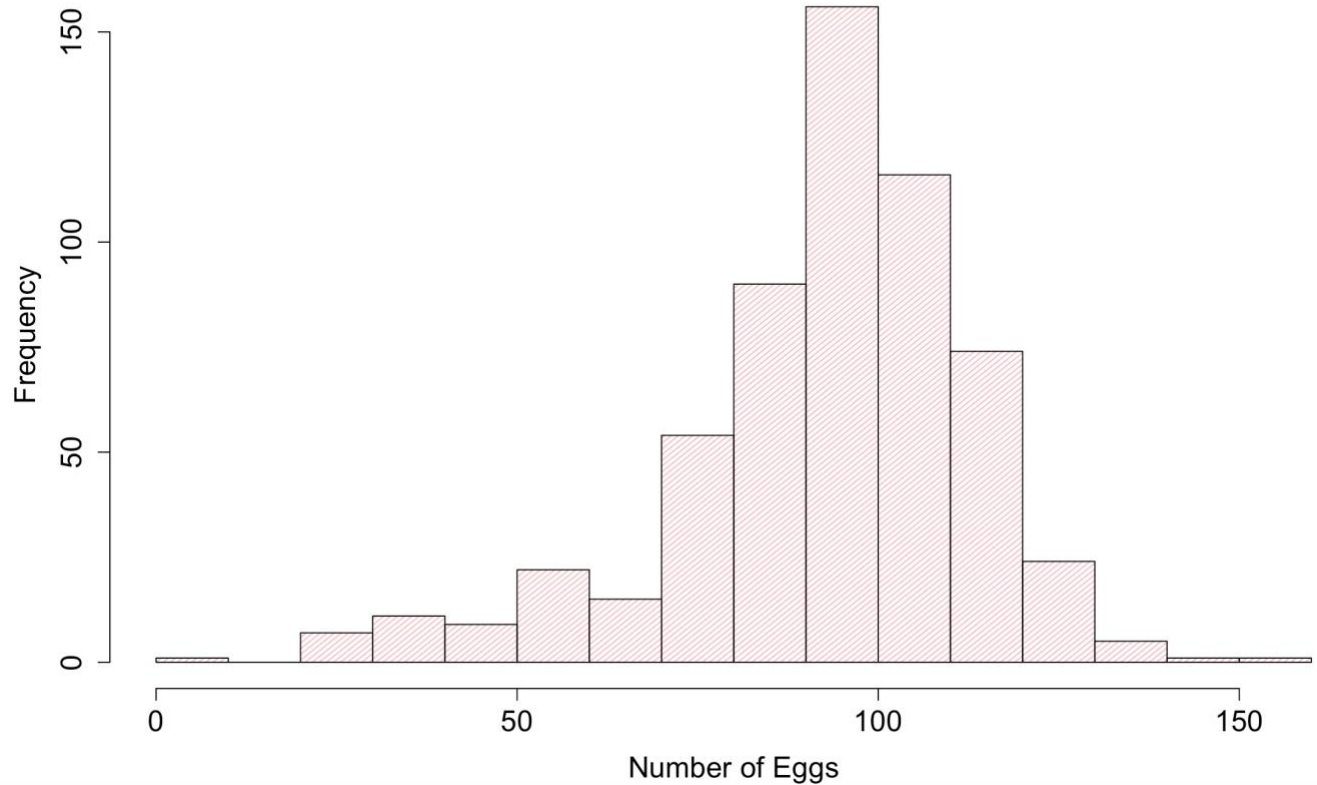


Figure 27: Clutch size distribution of excavated hatchery nests.

With measurements taken from a random sample of eggs from each nest for 122 nests, the average egg weight was 36.5 g (SD = 4.6 g); out of a random sample of 20 hatchlings from each nest for 101 nests, the average hatchling weight was 17.5 g (SD = 3 g); out of 117 nests, the average CCL was 4.4 cm (SD = 0.19 cm); and out of 116 nests, the average CCW was 4.5 cm (SD = 0.39 cm).

Any seasonal trends in the success of the hatchery could not truly be determined due to lack of data. Out of the three years of data collection, 2016 was the only year in which data on the success rate was collected for all months. The success rate of each month can be seen in Figure 28, but it is difficult to determine any trends with only one year of data.

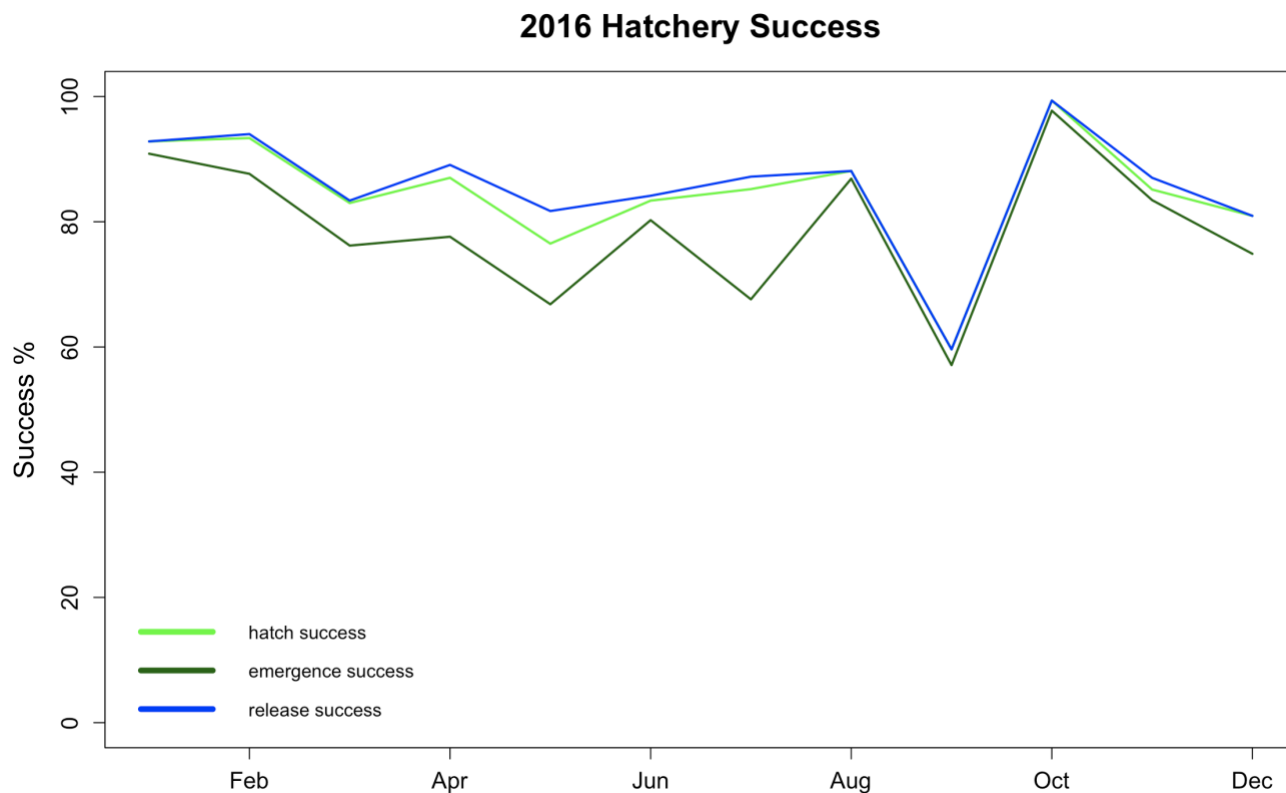


Figure 28: Average hatchery success rate by month in 2016.

DISCUSSION

NESTING PREFERENCES AND BIOLOGICAL DATA

Overall, the data collected at Piro and Playa Pejeperro support literature about nesting preferences and biological data on both the olive ridley and the green turtle. The mean CCL of the olive ridley was 66.5 cm, which falls in the typical range as documented by several studies of eastern Pacific populations: 63-73 cm (Aguirre et al. 1999); 60-69 cm (Angulo-Lozano et al. 2007); and 65-71 cm (Majewska et al. 2015). The mean CCL of the green turtle was 85.9 cm, which falls in the range of east Pacific populations documented by Blanco et al. 2012 (77.5 - 91.4 cm), but is smaller than the range of South Pacific populations (96.5-113 cm) (Craig et al. 2004). It is possible that some misidentification of species occurred during this study, resulting in olive ridleys with an abnormally high CCL and green turtles with an abnormally low CCL. The preferred nesting habitat of olive ridley turtles is a relatively flat area in the middle of the beach that is free of debris (NMFS and USFWS 1998), which is supported by the olive ridleys' preference for Zone 2 of the beach ($\chi^2 = 717.26$, $df = 2$, $p < 0.001$) (Figure 10). Many olive ridley nests are also found near low-salinity waters near river mouths, lagoons, and estuaries (Barquero-Edge, 2013; Ruckdeschel and Shoop, 2006), which is also supported by the documentation of nests at the Playa Piro river and Playa Pejeperro lagoon during the study period. Of the 13,130 nests for which distance to

vegetation was recorded, 13% were recorded at the lagoon at Playa Pejeperro or the river at Playa Piro. Seventy percent of these nests were olive ridley nests that were located at the lagoon ($n = 1181$). For the nesting adult females that were encountered, 8% were recorded at the river, lagoon, or cliff. Green turtles have been documented to prefer nesting in or near vegetation (Whitmore and Dutton 1985), which is supported by their preference for Zone 3 of the beach in this study ($\chi^2 = 717.26$, $df = 2$, $p < 0.000$) (Figure 10), as well as the lower distance to vegetation and higher distance to tide of adult female turtles as compared to olive ridley turtles (Tables 4 and 5, Figures 18 and 20).

I calculated the mean beach width per year to determine if there has been any change in the size of the beach over time, as researchers have stated that about 14 m has been lost in width during the study period (A. Whitworth, personal communication, October 1, 2018). The mean beach width (Figure 22) varied from year to year, with no significant loss or gain in the width of the beach over time (both beaches: $\chi^2=6$, $df=6$, $p = 0.423$). The natural variation in beach width was reflected in yearly variation in recorded distance to vegetation and distance to tide of adult nesting turtles (Figures 19 and 21). The tidal phase was not recorded during measurements. The tidal range on the Osa Peninsula is approximately 2.4 m (Lajoie 1986; Roy et al. 1994). This value is small enough in comparison to the mean distances that tidal phase would likely not make a difference when averaging the distance to tide measurement. This is an indirect system of estimating beach width and to truly investigate any instances of erosion and beach loss, more research is needed.

Trends among years involving the raw nest numbers are more difficult to determine because of the unequal effort both in terms of number of nights spent patrolling and time spent patrolling per year (Figure 2). For example, both the number of adult turtles encountered (Figure 16) and the number of nesting events at Playa Pejeperro (Figure 6) were lowest in 2017. It is possible that this reflects a decline in the number of nesting turtles, but it is more likely that the low numbers are a reflection of the low patrol effort that year at Playa Pejeperro (Figure 2). The hatching and emergence success of the wild nests across the study period appear to be relatively stable with natural variation (Figure 23), and the success of the hatchery is consistently high throughout the study (Figure 26). There were no discernable seasonal trends in success of either the wild nests or the hatchery (Figures 24 and 28).

HATCH SUCCESS

The hatch success of a clutch is typically calculated as the number of empty egg shells divided by the total number of eggs. Across the study period the annual mean hatch success (which in this study also included pipped live and pipped dead) ranged from 65.6% - 84.5%. This range is consistently higher than the typical hatch success that has been measured at mass nesting sites.

A study by Valverde et al. (2012) found that monthly mean hatching rates ranged from 0% to 32.6%. An earlier study by Cornelius and Robinson (1985) at Ostional also found an approximate success rate of 32%. At Playa Nancite, hatching success was found to be significantly affected by nest density (Honarvar and O'Connor 2008). High-density experimental nests had a hatching success of 29.5%, moderate-density nests a success of 55.9%, and low-density nests a success of 71.6% (Honarvar and O'Connor 2008). The high hatch success of wild nests in this study is consistent with the success rate of low-density nests in Honarvar and O'Connor (2008), supporting the theory that the low-density solitary olive ridley nests have higher hatchling emergence success than arribada nests and are important contributors to the olive ridley population (Dornfeld et al. 2015; Barquero-Edge 2013).

HATCHERY

Data collected from the hatchery also support literature on the typical clutch size, egg weight, and hatchling size and weight of olive ridley nests. Olive ridleys lay clutches of about 100 eggs (NMFS and USFWS 1998, National Research Council 2010). The mean clutch was 90.4 eggs (SD = 20.8) for the hatchery and 87.1 eggs (SD = 26.4) for wild nests. The clutch size is slightly lower than expected because these averages include data from partially predated nests. The average egg weight was 36.5 g (SD = 4.6 g), the average hatchling weight was 17.5 g (SD = 3 g), the average CCL was 4.4 cm (SD = 0.19 cm), and the average CCW was 4.5 cm (SD = 0.39 cm). These values all fall within an expected egg weight between 30 and 38 g (NMFS and USFWS 1998), a hatchling weight between 12 and 22.3 g (NMFS and USFWS 1998), and a CCL between 37.5 and 46 mm (Hirth 1980).

PREDATION

The most notable trends throughout the study period occur in relation to the predation data. The percentage of predated nests at both Playa Piro and Playa Pejeperro has been steadily increasing over the study period, reaching above 40% for both beaches in 2017 (Figure 11). Though Playa Piro and Playa Pejeperro are similar in most measures, their largest difference is in the dominant predators throughout the study period. When considering only nests predated by a single known predator, the top three predators at Playa Piro are coatis/raccoons (27.6% of nests), birds (15.4%), and dogs (11.2%) (Table 1, Figure 13). When taking into account all nests with known predators, including those predated by multiple predators, the top three predators remain the same but with higher percentages (Figure 13). The top three predators at Playa Pejeperro are dogs (29.9%), birds (16.4%), and coatis/raccoons (9.4%) (Figure 13). Again, when considering multiple predators, the top three predators remain the same, indicating high overlap of these predators with other predator types. This suggests that much of the predation, particularly by birds such as black vultures, is likely done as scavenging rather than direct nest destruction. In a study of avian predation on olive ridley eggs on Ostional Beach, vultures searched for hatchlings

on the surface, for eggs or hatchlings that were exposed by other predators, and for clusters of birds surrounding emerging hatchlings (Burger and Gochfeld 2014a). I hypothesize that a high level of predation by multiple predators occurs in this study area because predation of nests by scavengers is facilitated when eggs are exposed during initial predation events by species such as coatis and dogs, as well as by erosion of nests. However, in contrast to beaches with mass nesting events, predation on these beaches is not facilitated by the exposure of eggs by other nesting females (Burger and Gochfeld 2014a; M. Espinoza, personal communication, February 8, 2019).

The variation among years in the percentage of nests predated by each of these species is highly variable and inconsistent from year to year (Figures 14 and 15). Some predators stayed relatively consistent in their predation, such as humans and maggots, others were variable, such as dogs, and some increased over the study period, such as coatis/raccoons (Figure 14). Researchers have noticed an increase in the coati population in recent years (M. Espinoza, personal communication, February 8, 2019), but no published study has documented this increase in the region. It is likely that the increase in population is due to the low density and declining population of jaguars in Corcovado National Park (Salom-Perez et al. 2007), a species that eats coatis and raccoons but is considered “near threatened” in this region (Quigley et al. 2017). Jaguars experience both hunting pressure (Carrillo et al. 2000) and habitat loss due to deforestation (Sanchez-Azofeifa et al. 2002). Forest cover declined from 97% to 89% from 1979 to 1997 on the Osa Peninsula, indicating a loss in adequate habitat area for jaguars (Sanchez-Azofeifa et al. 2002). This loss of predation by top predators can cause an increase in the population of mid-level meso predators such as coatis, in turn causing an increase in sea turtle nest predation by these species.

COMPARISON TO PREVIOUS RESEARCH

There are three published studies on sea turtle nesting that have occurred on at least one or more of the beaches in this study: Drake (1993), Drake (1996), and Barquero-Edge (2013). Below is a summary of the results of each and how they compare to the results from this study.

Drake (1993)

The first quantitative study on marine turtle nesting on the Osa Peninsula occurred in 1993 in a preliminary assessment by Drake (Drake 1996). It is not specified which beach was surveyed, but only that it was a 4.2 km stretch 16 km east of Corcovado National Park. The study was conducted daily from June 18 – August 9, 1992, during which time a total of 135 nests were encountered. The focus of this study was predation rates and dominant predators. Drake found that 46.8% of nests were predated. Of the predated nests, dogs were responsible for 57%, humans for 18%, coatis for 7%, and 18% were unknown.

Drake (1996)

In 1996, Drake conducted a study on four beaches on the Osa Peninsula, including Playa Piro and Playa Pejeperro. Over a 14-month period, she collected data on spatial distribution of nests, the seasonality of nesting activity, the nest predation frequency, and the hatch frequency. This data serves as an excellent baseline for nesting data in this region as well as a suitable comparison for the data in this study, since data collection methods and analysis were very similar.

Drake determined that 97% of recorded nests were laid by olive ridleys and that nesting was highest in September and October. Drake also found that the nesting season of the green turtle was similar to the olive ridley. Of the total nests, 7% successfully hatched. Nests laid at the beginning of the rainy season (June) had greater hatch success than those laid at the beginning of the dry season (January) (Drake 1996).

Across the four beaches, 30% of nests were predated and rates were highest in September and October. Across all beaches, dogs were responsible for 49% of depredated nests, man was responsible for 42%, and coatis were responsible for 2%. It is important to note that predation categories were limited to these three predator types. Most depredation by coatis occurred on Playa Piro, and nest predation by man was lowest at Playa Piro. At Playa Pejeperro, human predation was greater than that of dogs (61% and 36%, respectively). At Playa Piro, predation by dogs was the highest at 47% of nests.

Barquero-Edge (2013)

Barquero-Edge monitored the same four beaches as Drake (Playa Piro, Playa Pejeperro, Playa Rio Oro, and Playa Carate) from July through December 2010. The purpose of this study was also to document the number of nests laid by each of the species, how nesting varies temporally and spatially, and the frequency of predation events. Barquero-Edge was assisted by volunteers from Osa Conservation and Frontier. The olive ridley turtle was the most abundant with 67% of nests. The highest number of nests were located at Pejeperro and the lowest at Rio Oro. Nesting for both the olive ridley and green turtle peaked in August and September. Of the 1,943 total nests, 39% were predated. Pejeperro had a predation rate of 50% while Piro had a predation rate of 47%. The most common predators of nests at both Piro and Pejeperro were dogs (36% and 52%, respectively). Coatis were the most common predator at Rio Oro and humans were the most common predator at Carate.

Summary

To best compare and contrast the results of my analysis, I will focus mainly on the results from Piro and Playa Pejeperro in these earlier studies. Olive ridleys laid 99% of nests on Playa Piro and

88% of nests on Playa Pejeperro in this study, which is consistent with the earlier studies determining olive ridleys as the most abundant species. When including false crawls, the percent of nesting events by olive ridleys decreases to 77% on Playa Piro and to 64% on Playa Pejeperro, supporting literature that states that green turtles leave higher number of false crawls than other species (Ruckdeschel and Shoop, 2006). Both Drake and Barquero-Edge found that nesting was highest between August and October, which is supported by the results of this analysis (Figure 7, Table 6). Both studies also found that the nesting season of the green turtle was similar to the olive ridley, but I found that peak nesting for the green turtle occurred from November - January and did not follow the same pattern as the olive ridley (Figure 7, Table 6).

Of the total nests, 7% successfully hatched, which is the same average hatch rate among all beaches calculated in Drake (1996). When limited to only Playa Piro and Playa Pejeperro, the average hatch rate in Drake (1996) was only 4%, indicating that the hatch rate at these two beaches has increased slightly (Table 6). Drake determined that nests laid at the beginning of the rainy season (June) had a greater hatch frequency than those laid at the beginning of the dry season (January), but I found no discernable seasonal trends in hatch success throughout the study period (Figure 24).

Drake found that predation rates were highest in September and October, but I found no monthly patterns in predation rate (Figure 12). When limited to only Piro and Playa Pejeperro, the predation rate averaged 35.5% for Drake's study and increased to 48.5% for Barquero-Edge. The mean predation rate between the two beaches in this study was 22% (Table 6). This indicates an overall decrease in predation, but when looking at individual years the predation rate reached almost 50% in the later years of the study. Thus, it is more likely that predation of nests is just as high, if not higher than when both of these studies were conducted. Since both of the earlier studies were short in duration it is also difficult to know whether the predation rate found was average, or abnormally high or low compared to previous or subsequent years. In addition, Barquero-Edge (with assistance from Osa Conservation) protected twelve nests on Playa Piro from predation with nest covers in 2010. Osa Conservation used nest covers in the past (M. Espinoza, personal communication, February 8, 2019) but it is unclear if 2010 was the only year or if the practice was continued into the early years of my analysis. If so, it may explain why predation was so low in the early years of this study (Figure 11).

The dominant predators on both beaches have changed since the previous studies (Table 6). The dominant predators in Drake (1996) were dog, man, and coati at Playa Piro (47%, 12%, and 12%, respectively) and man, dog, and coati at Playa Pejeperro (61%, 36%, and 1%). The dominant predators in Barquero-Edge (2013) were dog, coati, and human at Playa Piro (36%, 19%, and 10%, respectively) and dog, human, and coati at Playa Pejeperro (52%, 28%, and 4%). In this study,

27.6% of nests at Playa Piro were predated by coatis/raccoons, dogs predated only 11.2% of nests, and humans predated only 10.4% of nests (Figure 13, Table 1). At Playa Pejeperro 29.9% of nests were predated by dogs, 9.4% were predated by coati/raccoons, and only 5.8% were predated by humans. The predation categories in this study were also expanded to include birds, crabs, and maggots. Essentially, coatis/raccoons have emerged as the greatest threat to nests at Playa Piro and dogs remain the greatest threat at Playa Pejeperro. While predation by dogs has varied across the years of this study, the rise in predation of nests by coati over the years can be seen clearly in Figure 14. Poaching of nests by humans has decreased significantly for both beaches since the earlier studies, averaging only 8.2%. Though poaching is low at both beaches, it is still significantly higher at Playa Piro ($p < 0.0001$), which contrasts Drake’s finding that nest predation by man was lowest at Playa Piro because it is the least accessible and has the least human population of all the beaches studied.

Table 6: Comparison of this study to three previous sea turtle nesting studies on the Osa Peninsula. Results from earlier studies are limited to Playa Piro and Playa Pejeperro.

Study	Year(s)	Peak Nesting (Olive)	Peak Nesting (Green)	Hatch Success	Predation	Main Predators (Piro)	Main Predators (Pejeperro)
Drake (1993)	Jun - Aug 1992	N/A	N/A	N/A	46.8%	Dogs (57%) Humans (18%)	
Drake (1996)	Sep 1993 - Oct 1994	Sep-Oct	Sep-Oct	4%	35.5%	Dogs (47%) Humans (12%) Coati (12%)	Humans (61%) Dogs (19%) Coati (1%)
Barquero-Edge (2013)	Jul-Dec 2010	Aug-Sep	Aug-Sep	N/A	48.5%	Dogs (36%) Coati (19%) Humans (10%)	Dogs (52%) Humans (28%) Coati (4%)
This study	Jan 2011- Dec 2017	Aug-Oct	Nov-Jan	7%	22%	Coati (27.6%) Birds (15.4%) Dogs (11.2%)	Dogs (29.9%) Birds (16.4%) Coati (9.4%)

There are several possible reasons for the decline in poaching of eggs, such as the implementation of regular beach monitoring by Osa Conservation and other organizations, the rise of eco-tourism that provides jobs and income, or environmental education efforts. Anecdotally, researchers have described a trend in which poaching of eggs by humans increases during the low tourism season (July – December), because there is lower employment (A. Whitworth, personal communication, October 1, 2018). However, I did not find this pattern when looking at seasonal trends in predation (Figure 12). Barquero-Edge noted that local residents suggested that poachers may cover up excavated nests to hide their actions. It is possible that

volunteers unfamiliar with this action mark the predated nests as “unknown”, which may explain why the percentage of this nest classification is so high in that study (48%) as well as in this study (28.3%). If all “unknowns” in this study were actually human predation, then the rate would actually be 36.5%, which is similar to the rates documented in the earlier studies.

MANAGEMENT RECOMMENDATIONS

It is likely that threats to sea turtle nests and hatchlings in this region are the result of large anthropogenic actions, such as the removal of keystone predators and corresponding increase in the coati population. Osa Conservation cannot directly change these underlying problems, but there are actions that they as an organization can take to improve the important work they do in conserving nests. Osa Conservation has already collected an impressive amount of data and successfully implemented effective conservation strategies, so in my recommendations I aim to provide reasonable, practical steps they can take to strengthen their sea turtle monitoring program. My recommendations are as follows:

- 1) The continuation of monitoring and data collection in this region;
- 2) Expansion of monitoring to other beaches and/or data sharing among organizations;
- 3) Adoption of State of the World’s Sea Turtles (SWOT) protocol;
- 4) More defined predation categories;
- 5) The use of predator excluders to reduce predation;
- 6) The continuation of nest relocation to the hatchery with additional research on its effects

1) The continuation of monitoring and data collection in this region

Relatively little research has been published on the nesting trends in this region, despite the reproductive importance of solitary nesters for the larger population (Barquero-Edge, 2013). This study comprises the first analysis of multiple years of data collected at these beaches on the Osa Peninsula. I strongly encourage the continuation of monitoring and data collection on these two beaches in order to better understand nesting in this important region. If at all possible, I would encourage Osa Conservation to increase the use of volunteers in collecting nesting data on Playa Piro and Playa Pejeperro. With equal effort on both beaches and in following years, trends in the number of nests, the number of adult turtles encountered, and other data can actually be established as reflections of changes in the population and not as variation in effort.

2) Expansion of monitoring to other beaches and/or data sharing among organizations

Drake (1996) determined that Playa Rio Oro, adjacent to Playa Pejeperro, is the most important beach in terms of nesting activity. During Drake’s study, Playa Rio Oro had six times more nests than Playa Piro (1996). However, Playa Rio Oro had the lowest number of nests in Barquero-Edge

(2013) and Playa Carate had the second-highest number of nests behind Playa Pejeperro. It is possible that nesting females move among the different beaches and may favor specific beaches in specific years. Expanded monitoring of all beaches would elucidate the patterns of nesting for the larger region. If no other organizations are monitoring these beaches, I would encourage Osa Conservation to expand their monitoring. If other organizations are responsible for the monitoring of these beaches, I encourage data sharing among organizations to better establish data for the whole region, like the earlier studies, rather than limited to only two beaches.

3) Adoption of SWOT protocol

The State of the World's Sea Turtles (SWOT) database is the most comprehensive global sea turtle nesting database, relying on hundreds of data providers from more than 2800 nesting beaches (SWOT Scientific Advisory Board 2011). However, comparing data is difficult because there is no global standard of monitoring protocols that would allow data to be standardized. In 2011, the SWOT Scientific Advisory Board published a guide for the minimum data standards for both existing and new monitoring projects. I encourage Osa Conservation to adjust their monitoring protocol following this comprehensive guide, in order to achieve the highest ranking in the SWOT data classification system. Increasing monitoring effort to this level will improve the ability to detect trends in the nesting population.

Ideally, monitoring should occur at least three times per week throughout the nesting season, which will provide an acceptable level of confidence in the estimates of nesting abundance. If it is not possible to keep consistent monitoring effort across the study area, an alternative is to choose a single beach as an index for the population, assuming that all four beaches in the region are used by the same nesting population.

A small change in monitoring that would be easy to implement is counting all nesting activities during a monitoring event. This means that if monitoring occurs but no nesting attempts are recorded, a zero value should be included to account for lack of nesting activity. Including these values in addition to all other nesting activity will provide a more accurate picture of seasonal nesting activity in relation to monitoring.

Though Osa Conservation has standardized its monitoring in comparison to the earliest years of the study, there are still problems with consistent effort. Even if the goal is not to submit data to the SWOT database, this guide provides recommendations for maximizing effort seasonally as well as recommendations for analyzing data to best estimate seasonal abundance.

4) More defined predation categories

The implementation of defined predation categories in 2015 resulted in a decreased number of “unknowns” as compared to previous years where the predator was documented in a comment. However, strictly defined categories may result in some oversight of other predator types. For example, “ants” appeared as a predator on several comments in the raw data, but was never included as a predator category. Wetterer et al. (2007) found a high percentage of fire ants (*Solenopsis invicta*) in green turtle, leatherback, and loggerhead nests located close to vegetation, which poses a threat to eggs and hatchlings that can be negatively affected or killed by stings. It would be worthwhile to better understand if ants or other invertebrates are a problem in this area, particularly for green turtle nests that are typically located closer to vegetation.

In addition, I recommend splitting up the predation category of coati/raccoon into two separate predation categories. I assume they are categorized as one because of similar tracks, but distinguishing between the two, even if only through visual sighting, would provide insight as to whether the increased coati population really is leading to higher predation, or if other mid-level meso-predators are the problem. In Barquero-Edge (2013), coatis were identified by a small hole dug in the nest and prints that are 4.5 cm wide, which are more oblong and at least 1-1.5 cm smaller in width than raccoon prints. The use of wildlife cameras on the beach could also help verify the different mammalian predators and allow volunteers/staff to verify their classification of species by tracks only.

5) The use of predator excluders to reduce predation

During Barquero-Edge (2013) study, twelve nests on Playa Piro were protected with a 1-m square section of 10x10 wire netting and secured with stakes. The predation rate of the covered nests was 0%, compared to 100% of control nests that were predated. Netting has been successfully used to protect nests in other studies (Ali and Ibrahim 2002; Kurz et al. 2012; O'Connor et al. 2017; Ratnaswamy et al. 1997; Yerli et al. 1997), so I recommend that Osa Conservation re-implement the use of nest covers. It is possible that the magnetic environment from metal netting affects orientation and navigation behavior of hatchlings (Irwin et al. 2004), so I further recommend the use of other materials if possible. The use of bamboo covers to protect nests on nearby Carate Beach is documented as successful in Korein et al. (2019). Over one year, the implementation of bamboo covers used opportunistically when one was available resulted in a predation rate of 14.9%. Advantages of bamboo covers include being sturdy, lightweight, easy to move, simple to construct, inexpensive, allow hatchlings to emerge, and are more environmentally-friendly than plastic or metal if they wash out to sea (Korein et al. 2019).

In addition, predation by birds was a consistent threat at both beaches. While their predation is often facilitated by other predators, they are also a large threat to hatchlings during the crawl

time from the nest to the ocean, when it occurs during daylight hours (Burger and Gochfeld 2014b). While nest covers will likely decrease predation overall, avian predation can also be reduced by minimizing the crawl time for hatchlings after they emerge from the nest. This is best done by removing beach debris that can delay hatchling movement to the sea (Burger and Gochfeld 2014a). Though nighttime hatchling emergence is the most common behavior, keeping predatory birds off the beach during early morning and daytime hatchling emergence may also result in a greater percentage of hatchlings that successfully make it to the sea (Burger and Gochfeld 2014a).

6) The continuation of nest relocation to the hatchery with additional research on its effects

From 2015 to 2017, the relocation of nests to the hatchery resulted in consistently high hatching, emergence, and release success of turtles. I recommend that Osa Conservation continue the use of the hatchery each year, utilizing staff and volunteer effort to keep it well-managed and monitored. However, though egg relocation to protected hatcheries is a common practice, the impacts on hatchling recruitment and health have been brought into question. A study of the influence on thermal nest environment on emergence success and quality of olive ridley nests found that nest temperatures above 34°C for at least 3 consecutive days resulted in decreases in emergence success and locomotion of hatchlings (Maulany et al. 2012). In addition, temperature-dependent sex determination could result in sex ratio distortion of hatchlings in hatcheries with unregulated nest temperatures (Morreale 1982). The hatchery at Playa Piro is a great opportunity for Osa Conservation to research the potential impacts of the hatchery on sex ratio, emergence success of nests, locomotion, and other potential fitness impacts. A better understanding of these effects will help to ensure that their management techniques are not negatively impacting the hatchlings produced in the hatchery.

One potential addition to data collection in the hatchery is the incorporation of shade data. When relocating nests to the hatchery, researchers place eggs in either a shaded or non-shaded part of the hatchery (A. Whitworth, personal communication, October 1, 2018). Green turtle nests are automatically placed in the shaded area (A. Whitworth, personal communication, October 1, 2018) because temperature affects both the sex ratio and body size of hatchlings (Glen et al. 2003). I suggest that the placement of a nest in either shade or no shade be documented in the data, so its effect on success and the size and weight of hatchlings can be studied further. Researching the potential impacts of shading on hatching success, size, behavior, and sex ratio will provide a better understanding of the effect of this management technique on hatchlings.

CONCLUSION

Studies on the beaches in this region are limited, despite their importance as a solitary nesting habitat for olive ridley turtles. The data collected during this study provide an update to previous

monitoring studies conducted in this region, as well as serve as baseline data with which to compare future studies. Overall, the data support literature about nesting preferences and biological data on both the olive ridley and the green turtle. Both beaches are similar in species composition, nesting frequency, hatch success, and predation. The greatest difference between the two is the dominant predator types, with Playa Piro experiencing greater predation by coatis/raccoons, and Playa Pejeperro experiencing greater predation by dogs. The predation percentage and dominant predator types is also the greatest difference from earlier studies published by Drake (1996) and Barquero-Edge (2013). Overall predation of nests still remains a significant threat on both beaches, though poaching of eggs by humans has decreased. The relocation of eggs to the hatchery at Playa Piro results in high hatch and emergence success, but more research is needed on potential impacts to the hatchlings.

This research will help Osa Conservation improve their monitoring and management strategies regarding nesting, as well as contribute to better understanding of nesting trends in this important region. I encourage the organization to continue their monitoring of these beaches, and if possible to expand to the adjacent beaches in the region and increase their monitoring effort. Reference to the SWOT protocol may help the organization maximize and standardize their effort, allowing studies to be conducted on the viability of this population of nesting turtles. Predation is the greatest threat to nests, but can be better understood through slight changes to data collection. I highly encourage the covering of nests with predator exclosures, which will likely greatly decrease predation by both wild and domesticated animals. Lastly, the relocation of threatened nests to the hatchery has proved very successful, so I encourage Osa Conservation to continue this conservation strategy while incorporating research projects to assess the impact of incubation on hatchlings. In addition, I suggest that they conduct research on the efficacy of these management techniques to better understand their effect on nest preservation and hatchling recruitment.

ACKNOWLEDGEMENTS

This project would not be possible without the generous help and support of my colleagues, friends, and family. I would like to thank my advisors Dr. Matthew Godfrey and Dr. Lisa Campbell. In particular, Dr. Godfrey's guidance and wealth of sea turtle knowledge throughout this process ensured my success, and I thank him for being available and supportive despite the distance between Durham and Beaufort. I would also like to thank staff at Osa Conservation: Dr. Andy Whitworth for introducing me to and allowing me to join this project; Marina Garrido for walking me through all the data prior to my analysis; Monica Espinoza and Noelia Hernandez for answering my many questions; and Manuel Sanchez for coordinating and organizing this project over many years. I am inspired by the hard work of your organization and your dedication to

conservation. Finally, thank you to Kimberley Drouin for helping me with my analysis, and my family and friends for their endless support over these past two years.

REFERENCES

- Abreu-Grobois, A & Plotkin, P. (IUCN SSC Marine Turtle Specialist Group) 2008. *Lepidochelys olivacea*. *The IUCN Red List of Threatened Species* 2008: e.T11534A3292503. <http://dx.doi.org/10.2305/IUCN.UK.2008.RLTS.T11534A3292503.en>. Downloaded on 13 March 2019.
- Aguirre, A. A., Spraker, T.R., Chaves, A., Du Toit, L., Eure, W., Balazs, G.H. (1999). Pathology of Fibropapillomatosis in Olive Ridley Turtles *Lepidochelys olivacea* Nesting in Costa Rica. *Journal of Aquatic Animal Health*, *11*, 283-289.
- Ali, A., Ibrahim, K. (2002). *Crab predation of green turtle (Chelonia mydas) eggs incubated on a natural beach and in turtle hatcheries*. Paper presented at the Proceedings of the 3rd Workshop on SEASTAR2000.
- Angulo-Lozano, L., Nava-Duran, P.E., Frick, M.G. (2007). Epibionts of Olive Ridley Turtles Nesting at Playa Ceuta, Sinaloa, Mexico. *Marine Turtle Newsletter*, *118*, 13-14.
- Ankersen, T. T., Regan, K.E., Mack, S.A. (2006). Towards a bioregional approach to tropical forest conservation: Costa Rica's Greater Osa Bioregion. *Futures*, *38*, 406-431.
- Antworth, R. L., Pike, D.A., Stiner, J.C. (2006). Nesting ecology, current status, and conservation of sea turtles on an uninhabited beach in Florida, USA. *Biological Conservation*, *130*, 10-15.
- Barquero-Edge, P. S. (2013). Trends in marine turtle nesting and egg predation on the Osa Peninsula, Costa Rica. *Marine Turtle Newsletter* (138), 7-10.
- The Biology of Sea Turtles. (1997). (P. L. Lutz, Musick, J.A. Ed. Vol. 1): CRC Press.
- Blanco, G. S., Morreale, S.J., Bailey, H., Seminoff, J.A., Paladino, F.V., Spotila J.R. (2012). Post-nesting movements and feeding grounds of a resident East Pacific green turtle *Chelonia mydas* population from Costa Rica. *Endangered Species Research*, *18*, 233-245.
- Burger, J., Gochfeld, M. (2014). Avian Predation on Olive Ridley (*Lepidochelys olivacea*) Sea Turtle Eggs and Hatchlings: Avian Opportunities, Turtle Avoidance, and Human Predation. *Copeia*(1), 109-122.

- Burger, J., Gochfeld, M. (2014). Factors Affecting Locomotion in Olive Ridley (*Lepidochelys olivacea*) Hatchlings Crawling to the Sea at Ostional Beach, Costa Rica. *Chelonian Conservation and Biology*, 13(2), 182-190.
- Carrillo, E., Wong, G., Cuaron, A.D. (2000). Monitoring mammal populations in Costa Rican protected areas under different hunting restrictions. *Conservation Biology*, 14(6), 1580-1591.
- Cornelius, S. (1985). Update on Ostional. *Marine Turtle Newsletter*, 33, 5-8.
- Craig, P., Parker, D., Brainard, R., Rice, M., Balazs, G. (2003). Migrations of green turtles in the central South Pacific. *Biological Conservation*, 116, 433-438.
- Drake, D. L. (1993). Osa Sea Turtle Study. *Marine Turtle Newsletter*, 61, 9-11.
- Drake, D. L. (1996). Marine Turtle Nesting, Nest Predation, Hatch Frequency, and Nesting Seasonality on the Osa Peninsula, Costa Rica. *Chelonian Conservation and Biology*, 2(1), 89-92.
- Dornfeld, T. C., Robinson, N.J., Santidrian Tomillo, P., Paladino, F.V. (2015). Ecology of solitary nesting olive ridley sea turtles at Playa Grande, Costa Rica. *Marine Biology*, 162, 123-139.
- Girondot, M., Tucker, A.D., Rivalan, P., Godfrey, M.H., Chevalier, J. (2002). Density-dependent nest destruction and population fluctuations of Guianan leatherback turtles. *Animal Conservation*, 5, 75-84.
- Glen, F., Broderick, A.C., Godley, B.J., Hays, G.C. (2003). Incubation environment affects phenotype of naturally incubated green turtle hatchlings. *Journal of the Marine Biological Association of the United Kingdom*, 83(5), 1183-1186.
- Frazer, N. B., Ladner, R.C. (1986). A Growth Curve for Green Sea Turtles, *Chelonia mydas*, in the U.S. Virgin Islands, 1913-14. *Copeia*, 1986(3), 798-802.
- Hart, C. E., Ley-Quinonez, C., Maldonado-Gasca, A., Zavala-Norzagaray, A., Alberto Abreu-Grobois, F. (2014). Nesting characteristics of olive ridley turtles (*Lepidochelys olivacea*) on El Naranjo Beach, Nayarit, Mexico. *Herpetological Conservation and Biology*, 9(2), 524-534.
- Heithaus, M. R. (2013). In J. Wyneken, Lohmann, K.J., Musick, J.A. (Ed.), *The Biology of Sea Turtles* (Vol. III, pp. 475): CRC Press.

- Hirth, H. F. (1980). Some aspects of the nesting behavior and reproductive biology of sea turtles. *American Zoology*, 20, 507-523.
- Honarvar, S., O'Connor, M. P., & Spotila, J. R. (2008). Density-dependent effects on hatching success of the olive ridley turtle, *Lepidochelys olivacea*. *Oecologia*, 157(2), 221-230.
- Irwin, W. P., Lohmann, K.J. (2005). Disruption of magnetic orientation in hatchling loggerhead sea turtles by pulsed magnetic fields. *Journal of Comparative Physiology A*, 191, 475-480.
- James, R., Melero, D. (2015). Nesting and conservation of the Olive Ridley sea turtle (*Lepidochelys olivacea*) in playa Drake, Osa Peninsula, Costa Rica (2006-2012). *Revista de biologia tropical*, 63, 117-129.
- Korein, E., Caballol, A., Lovell, P., Exley, L., Porras Marin, C., Carrillo, J., Bond, G. Capria, L., Earl, S., Ferrari, O.M., Hamm, J., Johnson-Gutierrez, S., King, C., Malmierca, A., McAnally, L. Price, E. Riddick, E., Stokes, L. (2019). Using Bamboo Nest Covers to Prevent Predation on Sea Turtle Eggs. *Marine Turtle Newsletter*, 156, 33-37.
- Kurz, D. J., Straley, K.M., DeGregorio, B.A. (2012). Out-foxing the red fox: how best to protect the nests of the Endangered loggerhead marine turtle *Caretta caretta* from mammalian predation? *Oryx*, 46(2), 223-228.
- Lajoie, K. R. (1986). Coastal tectonics, in *Active Tectonics*, Stud. Geophys., pp. 95 – 124, Geophys. Study Comm., Natl. Acad. Sci., Washington D. C
- Majewska, R., Santoro, M., Bolanos, F., Chaves, G., De Stefano, M. (2015). Diatoms and Other Epibionts Associated with Olive Ridley (*Lepidochelys olivacea*) Sea Turtles from the Pacific Coast of Costa Rica. *PLOS One*.
- Maulany, R. I., Booth, D.T., Baxter, G.S. (2012). The effect of incubation temperature on hatchling quality in the olive ridley turtle, *Lepidochelys olivacea*, from Alas Purwo National Park, East Java, Indonesia: implications for hatchery management. *Marine Biology*, 159, 2651-2661.
- Morreale, S. J. (1982). Temperature-Dependent Sex Determination: Current Practices Threaten Conservation of Sea Turtles. *Science*, 216, 1245-1247.
- Mortimer, J.A & Donnelly, M. (IUCN SSC Marine Turtle Specialist Group) 2008. *Eretmochelys imbricata*. *The IUCN Red List of Threatened Species* 2008: e.T8005A12881238. <http://dx.doi.org/10.2305/IUCN.UK.2008.RLTS.T8005A12881238.en>. Downloaded on 13 March 2019.

The National Academies of Sciences, Engineering, and Medicine. 2017. *Effective Monitoring to Evaluate Ecological Restoration in the Gulf of Mexico*. Washington, DC: The National Academies Press.

National Marine Fisheries Service and U.S. Fish and Wildlife Service. 1998. *Recovery Plan for U.S. Pacific Populations of the Olive Ridley Turtle (Lepidochelys olivacea)*. National Marine Fisheries Service, Silver Spring, MD.

National Research Council 2010. *Assessment of Sea-Turtle Status and Trends: Integrating Demography and Abundance*. Washington, DC: The National Academies Press.

O'Connor, J. M., Limpus, C.J., Hofmeister, K.M., Allen, B.L., Burnett, S.E. (2017). Anti-predator meshing may provide greater protection for sea turtle nests than predator removal. *PLOS One*, 12(2).

Oros, J., Torrent, A., Calabuig, P., Deniz, S. (2005). Diseases and causes of mortality among sea turtles stranded in the Canary Islands, Spain (1998-2001). *Diseases of Aquatic Organisms*, 63, 13-24.

Osa Conservation. (n.d.). *Mission & Vision*. Retrieved February 22, 2019 from <http://osaconservation.org/mission-vision/>

Polovina, J. J., Balazs, G.H., Howell, E.A., Parker, D.M., Seki, M.P., Dutton, P.H. (2004). Forage and migration habitat of loggerhead (*Caretta caretta*) and olive ridley (*Lepidochelys olivacea*) sea turtles in the central North Pacific Ocean. *Fisheries Oceanography*, 13(1), 36-51.

Quigley, H., Foster, R., Petracca, L., Payan, E., Salom, R. & Harmsen, B. 2017. *Panthera onca* (errata version published in 2018). *The IUCN Red List of Threatened Species 2017*: e.T15953A123791436. <http://dx.doi.org/10.2305/IUCN.UK.2017-3.RLTS.T15953A50658693.en>. Downloaded on 23 February 2019.

Ratnaswamy, M. J., Warren, R.J., Kramer, M.T., Adam, M.D. (1997). Comparisons of lethal and nonlethal techniques to reduce raccoon depredation of sea turtle nests. *The Journal of Wildlife Management*, 61(2), 368-376.

Roy, P. S., P. J. Cowell, M. A. Ferland, and B. G. Thom (1994). Wave-dominated coasts, in *Coastal Evolution: Late Quaternary Shoreline Morphodynamics*, edited by R. W. G. Carter and C. D. Woodroffe, p. 121 – 187, Cambridge Univ. Press, New York

- Ruckdeschel, C., Shoop, C.R. (2006). *Sea Turtles of the Atlantic and Gulf Coasts of the United States*: University of Georgia Press.
- Salom-Perez, R., Carrillo, E., Saenz, J.C., Mora, J.M. (2007). Critical condition of the jaguar *Panthera onca* population in Corcovado National Park, Costa Rica. *Oryx*, 41(1), 51-56.
- Sanchez-Azofeifa, G. A., Rivard, B., Calvo, J., Moorthy, I. (2002). Dynamics of Tropical Deforestation Around National Parks: Remote Sensing of Forest Change on the Osa Peninsula of Costa Rica. *Mountain Research and Development*, 22(4), 352-358.
- Seminoff, J.A. (Southwest Fisheries Science Center, U.S.) 2004. *Chelonia mydas*. *The IUCN Red List of Threatened Species 2004*: e.T4615A11037468. <http://dx.doi.org/10.2305/IUCN.UK.2004.RLTS.T4615A11037468.en> . Downloaded on 13 March 2019.
- SWOT Scientific Advisory Board. 2011. *The State of the World's Sea Turtles (SWOT) Minimum Data Standards for Nesting Beach Monitoring, version 1.0*. Handbook, 28 pp.
- Valverde, R. A., Orrego, C. M., Tordoir, M. T., Gómez, F. M., Solís, D. S., Hernández, R. A., . . . Spotila, J. R. (2012). Olive Ridley Mass Nesting Ecology and Egg Harvest at Ostional Beach, Costa Rica. *Chelonian Conservation and Biology*, 11(1), 1-11.
- Wallace, B.P., Tiwari, M. & Girondot, M. 2013. *Dermochelys coriacea*. *The IUCN Red List of Threatened Species 2013*: e.T6494A43526147. <http://dx.doi.org/10.2305/IUCN.UK.2013-2.RLTS.T6494A43526147.en>. Downloaded on 13 March 2019.
- Wetterer, J. K., Wood, L.D., Johnson, C., Krahe, H., Fitchett, S. (2007). Predaceous Ants, Beach Replenishment, and Nest Placement by Sea Turtles. *Environmental Entomology*, 36(5), 1084-1091.
- Whitmore, C. P., Dutton, P.H. (1985). Infertility, embryonic mortality and nest-site selection in leatherback and green sea turtles in Suriname. *Biological Conservation*, 34(3), 251-272.
- Yerli, S., Canbolat, A.F., Brown, L.J., Macdonald, D.W. (1997). Mesh grids protect loggerhead turtle *Caretta caretta* nests from red fox *Vulpes vulpes* predation. *Biological Conservation*, 82, 109-111.