Interactions of Peruvian small scale fisheries with threatened marine vertebrate species

Submitted by Jeffrey Charles Mangel to the University of Exeter
as a thesis for the degree of
Doctor of Philosophy in Biological Sciences
In February 2012

I certify that all material in this thesis which is not my own work has been identified and that no material has previously been submitted and approved for the award of a degree by this or any other University.

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Abstract

Marine vertebrate species face unprecedented and ever increasing pressures as a result of human activity, primarily fishing, in the global oceans. One area of growing concern has been for the impacts of small-scale fisheries on these species. Over the past decade it has become increasingly clear that these under-studied fisheries have sizeable levels of catch and bycatch of many threatened and endangered species of sea turtles, seabirds and small cetaceans. This thesis presents a collection of chapters that investigate aspects related to the interactions of small-scale fisheries with threatened marine vertebrates.

We identify sizeable rates of bycatch of small cetaceans and seabirds for multiple small-scale longline and gillnet fisheries in both Peru and Ecuador. Catch rates of small cetaceans by the Peruvian small-scale driftnet fleet are estimated to exceed 10,000 dolphins and porpoises annually. A trial of acoustic alarms (pingers) in this same fishery showed a 37% reduction in small cetacean bycatch while not reducing target catch and represents a promising bycatch mitigation measure. Seabird bycatch was also found to be high in both longline and gillnet fisheries and included a wide range of seabird species including the critically endangered waved albatross (*Phoebastria irrorata)*. Through post-capture satellite tracking of loggerhead turtles (*Caretta caretta*) we show that these turtles are present in pelagic waters off the coasts of Peru and Chile for extended periods during which they are at risk of repeat interactions with small-scale longline fisheries operating throughout their foraging habitat. Through scan and focal sampling of the endangered marine otter (*Lontra felina*) we also show that otters making den sites in human fishing communities face additional risks due to entanglement in fishing gear or interactions with feral animals but, if properly managed, these sites could serve as stepping stones for marine otters along the coast.

The results presented here, gathered using a wide range of techniques, including onboard observer and shore-based monitoring, satellite tracking, bycatch quantification, and bycatch mitigation experiments, represent an attempt to better characterize and quantify the interactions of small-scale fisheries with threatened marine vertebrates toward identifying solutions that can lead to sustainable fisheries and populations of these protected marine species.

List of Contents

Acknowledgements

This PhD was made possible by the financial support of the Overseas Research Student Awards Scheme and the University of Exeter to whom I am most grateful.

I would also like to extend my sincere thanks and appreciation to all of the following: To my advisors Brendan and Stuart for their unwavering support and encouragement and endless fount of good ideas. To my fellow University of Exeter students, Matt Witt and Alan Rees in particular, with whom I have shared this time, and on more than one occasion, their desks. To all my colleagues at ProDelphinus - Paola Melly, David Montes, Mariela Pajuelo, Celia Caceres, Natalia Ortiz, Nadia Balducci, Andrea Luna and Eliana Alfaro - for their dedication and hard work that has helped make these studies possible. To the incredible and dedicated fishermen with whom we work - especially Miguel Cuentas, 'Chaval' Bernedo, Raul Beltran and Mateo Mamani. To my friends, mentors and colleagues in conservation who through the years have guided, advised and inspired me, Bill Gilmartin, Hannah Bernard, Jeff Seminoff, Peter Dutton, Kim Rivera, Janice Molloy, Rebecca Rose, Jessica Hardesty and Nigel Brothers. To the organizations that have had faith in us and provided the long-term support that has made this work possible: National Oceanic and Atmospheric Administration, National Fish and Wildlife Foundation, Columbus Zoo and Aquarium, Darwin Initiative and Agreement on the Conservation of Albatrosses and Petrels.

To my parents, in-laws, and brother who have supported both Joanna and I immensely throughout this process. To Tina, who keeps our house running and our boys happy, fed, and on-time. To my boys Kenzo and Jake for their patience and understanding through many weeks of travel and late nights and weekends at work. And most of all to Joanna, my wife, colleague and fellow student for her constant, unwavering support, I could not and would not have done this without you by my side.

List of Tables and Figures

Chapter I

Post-capture movements of loggerhead turtles in the southeastern Pacific Ocean assessed by satellite tracking

Table 1. *Caretta caretta*. Summary of environmental and tracking variables for 14 loggerhead turtles captured as bycatch off the coast of Peru. Ports are Ilo (I) and Pucusana (P). PTT models SPOT5 and SDR-T16 were obtained from Wildlife Computers while the remainder (ST-18, ST-20, ST-20D, A1010) were from Telonics. Attachment methods referred to are using fiberglass resin (F) or Powerfast epoxy (P). Duty cycle 1 was a laddered program set to transmit 24 h on (month 1), 6 h on and 13 h off (months 2 to 3), and 6 h on and 25 h off (>month 3). Duty cycle 2 was a laddered program set to transmit 24 h on (month 1), 24 h on and 48 h off (month 2 to 3), 24 h on and 72 h off (month 4 to 5), and 24 h on and 96 h off (>month 6). Injury score 1: external injuries (including entanglement); injury score 2: minor internal injuries to mouth or lower mandible; injury score 3: severe injuries due to deep hooking. D refers to displacement (Columns 10 and 11). CCL = curved carapace length

Figure 1. *Caretta caretta*. Map of all turtle track locations (60+ day turtles) by level of injury and showing a polygon of longline fishing effort monitored from eight ports (242 trips, 1771 sets) collected by fisheries observers from 2000 to 2007 (Alfaro-Shigueto et al. 2008). Gray shading of tracks indicates injury scores: light gray = injury score 1 (n=7), dark gray = injury score 2 (n=4), black = injury score 3 (n=3). The termination points of tracks of injury scores 2 and 3 are also marked with shaded squares and triangles, respectively. Tracked loggerhead positions were within fishing area boundaries from 75% ± 33% of the time (Range 13% to 100%) (250 m, 750 m, 2000 m, and 3000 m bathymetric contours are also shown). Inset map shows the predominant current patterns (arrows) of the southeastern Pacific Ocean. MCP: minimum convex polygon

Figure 2. *Caretta caretta.* Pooled monthly SST for all turtles tracked for >60 d (n = 11).

Figure 3. *Caretta caretta.* Track durations, minimum overall average swim speeds, displacement rates and time within fishing grounds minimum convex polygon (MCP) grouped by turtle size (curved carapace length, CCL) and injury score.

Figure S1. *Caretta caretta*. All turtle track locations (60+ day turtles) by level of injury and showing a polygon of longline fishing effort monitored from eight ports (242 trips, 1771 sets) collected by fisheries observers from 2000 to 2007 (Alfaro-Shigueto et al. 2008). Color shading of tracks indicates injury scores: red = injury score 1 (n=7), blue = injury score 2 (n=4), green = injury score 3 (n=3). The termination points of tracks of injury scores 2 and 3 are also marked with colored squares and triangles, respectively. Tracked loggerhead positions were within fishing area boundaries from 75% ± 33% of the time (Range 13% to 100%), (250 m, 750 m, 2000 m, and 3000 m bathymetric contours are also shown). Inset map shows the predominant current patterns (arrows) of the southeastern Pacific Ocean. MCP: minimum convex polygon.

Figure S2. *Caretta caretta*. Number of uplinks received per day for each turtle for last 45 days of the track. X-axis is days until track termination beginning at day 45. There is no evidence for a spike in uplinks toward the end of any track which could indicate that a turtle was floating injured or dead at the ocean surface. A change in transmitter duty cycle can be clearly noted in tracks CC10 and CC11 at approximately the 30 d mark.

Figure S3. *Caretta caretta.* Scatterplots showing the relationship of distance of release location from capture location with (a) maximum displacement ($r^2 = 0.0053$; n = 11), (b) total distance traveled after 1 mo ($r^2 = 0.0003$; n = 10), and (c) closest approach to capture location ($r^2 = 0.436$; n = 11).

Figure S4. *Caretta caretta.* Daily displacements grouped by Injury scores 1, 2 and 3 (Panels a, b and c, respectively).

Chapter II

Latitudinal variation in diet and patterns of human interactions in the marine otter

Table 1. Marine otter prey composition at Morro Sama (MS) and Vila Vila (VV), Peru based upon focal follow observations (MS *n*=513; VV *n*=211).

Table 2. Frequency (%) of marine otter interactions with humans, fisheries, and other species (*n*=190) as observed during 1,392 h of focal follows.

Figure 1. Locations of field sites for this study distributed along approximately 2,400 km of coastline in Peru and Chile. The upper inset map shows the five Peru study sites while the lower inset map shows the project area extent (black square) and the entire range of marine otters (in gray).

Figure 2. Diet composition given in relative frequency (RF%) by crustaceans and fish prey, found in fecal samples of marine otters collected in southern Peru (A) and four study sites in Chile (B to E). Sample size is shown for each sampled site.

Figure 3. Habitat use by observation zones for Morro Sama and Vila Vila based upon scan sampling observations. Bars indicate the percent of total observation time otters were seen in zones grouped by level of human presence (low, medium, high).

Chapter III

Small cetacean captures in Peruvian artisanal fisheries: High despite protective legislation

Table 1. Species composition, capture methods and use of small cetacean carcasses of all interactions (n[%]) with gillnet and longline vessels. Percentages are read across for capture methods and uses while species composition of gillnet bycatch subtotal and grand total (2nd and last columns from left) are tallied by column. The fate category "unknown" refers to animals for which final fate was not recorded.

Table 2. Estimated annual bycatch of small cetaceans by gillnet vessels for the port of Salaverry for the years 2002 - 2007, mean (CI). Values are derived from annually pooled monthly estimates of bycatch and known levels of monthly fishing effort for the port. Presented estimates are of total estimated small cetacean captures and of the four most commonly captured species.

Table S1. Summary of monitored fishing gears and their characteristics. Mean ± SD (Range).

Table S2. Gillnet total catch (bycatch and harpoon combined) and bycatch CPUE of all small cetaceans combined and of the four most commonly captured species. To facilitate comparison with other studies results are presented as catch. trip^{-1}, catch .set^{-1}, catch.km of net^{-1}, and catch .km^{2} of net area^{-1} but see Methods section 2.5 for a detailed description of the procedure for estimating total bycatch presented in Table 2. Values are averages of pooled monthly catch and bycatch CPUE observed during the study period, March 2005 - July 2007. Total catch estimates are not presented per set, per km and per km^{2}, because harpooning occurs independent of fishing effort at the set level.

Figure 1. Set locations by (a) gillnet vessels, (b) longline vessels, and (c) of all gillnet bycatch. Also presented in each pane are 50% and 75% probability contours of fishing sets and gillnet bycatch (250m, 750m, 2,000m and 3,000m isobaths are indicated).

Figure 2. Monthly average number of trips by gillnet and longline vessels for the years 2002 - 2007 determined from daily dock-side monitoring of fishing activity.

Figure S1. Set locations of gillnet bycatch of (a) *Delphinus* spp., (b) *L. obscurus*, (c) *T. truncatus,* and (d) *P. spinipinnis*. Also presented in each pane are 50% and 75% probability contours of fishing sets and gillnet bycatch (250m, 750m, 2,000m and 3,000m isobaths are indicated).

Figure S2. Mean monthly gillnet bycatch.set^{-1} of (a) total small cetaceans, (b) *Delphinus* spp., (c) *L. obscurus*, (d) *T. truncatus,* and (e) *P. spinipinnis*. Data are pooled by month for the study period of March 2005 - July 2007.

Chapter IV

Using pingers to reduce small cetacean bycatch in the small-scale driftnet fishery in Peru

Table 1. Gear characteristics and fishing effort for control (no pinger) and experimental (pinger) fishing sets observed during the study, April 2009 to August 2011. Mean ± SD (Range).

Table 2. Species composition of small cetacean bycatch for control (no pinger) and experimental (pinger) fishing sets. Also shown is the final use of the carcasses, presented as percentage, by species, within each of the given categories. "Discarded" refers to animals recovered dead and discarded at sea. "Bait" refers to animals recovered dead and then butchered for use as bait during subsequent fishing sets. "Food" refers to animals recovered dead and then butchered for food either for use during the trip or for home consumption.

Table 3. Catch rates of bycatch (dolphins and porpoises) and target catch (sharks and rays) for all observed control (no pinger) and experimental (pinger) sets, April 2009 to August 2011. Catch rates are presented as catch $(km^2h)^{-1}$ and are derived from the generalized linear mixed models used to test pinger effectiveness. Values shows are the mean and range ± 1 standard error (SE). Also shown is the percent change in catch rate between control and experimental sets and associated p-values.

Figure 1. Locations of control (filled circles) and experimental (open circles) fishing sets observed over the 29 months of the study (April 2009 to August 2011). Fishing vessels participating in the study were based in the port of Salaverry.

Figure 2. Locations and quantities of small cetacean bycatch in control (panel a) and experimental sets (panel b). The shaded area is the minimum convex polygon of fishing sets for each.

Table S1. Per set catch rates for all small cetacean bycatch species by control (no pinger) and experiment (pinger) treatment groups.

Figure S1. Locations and quantities of shark and ray catch in control (panel a) and experimental sets (panel b).

Figure S2. Bar-plots with standard error bars comparing the bycatch rates (catch set-1) for the main four small cetacean species for control (no pinger) and experimental sets (pinger).

Chapter V

Onboard observer data suggest that small-scale fisheries are a major potential threat to seabirds in the southeastern Pacific

Table 1. Summary of fisheries and observer effort by country. Port abbreviations are Santa Rosa (SR), Anconcito (An), Ilo and Salaverry (Sal). Fleet sizes are for these specific ports.

Table 2. Summary by country and fishery of the mortality rates, capture times, locations of hooking or entanglement and final uses of all seabirds captured during the study. Fisheries are listed by gear type (longline or net) and their main target species. Observed seabird catch per unit effort (CPUE; mean and 95% CI) by country, fleet and target species. Catch per unit effort is presented per 1000 hooks for longline fisheries and per km^2h for the driftnet fishery. Hook location categories are beak (Be), throat (Th), chest (Ch), wing (Wi) and other (Oth). Bycatch use categories are released alive (Rel), discarded dead (Dis), eaten onboard (Eat) and other (Oth). Seabird type codes are black-browed albatross (BBAL), waved albatross (WVAL) and white-chinned petrel (WCPE).

Table 3. Annual seabird catch estimates for observed fisheries and noting particular species of concern given their frequency as bycatch or their regional or global conservation status. Annual effort and observed catch rates (CPUE) are derived from this study or from published reports of fleet-wide fishing effort. Sources of national level fishing effort data for Peruvian fisheries are: Alfaro-Shigueto et al. 2010; Escudero 1997; Estrella 2007; Estrella et al. 1999; Estrella et al. 2000. Effort data for Ecuadorian fisheries were obtained during this study. Hake fishery data are national level while tuna fishery data are for the port of Santa Rosa only. Values in square brackets are extrapolations to allow for calculation of catch estimates based upon the available metrics of fishing effort.

Figure 1. Fisheries sampled and locations and relative quantities of seabird bycatch. Shaded areas are minimum convex polygons (MCP) of locations of observed fishing sets for the monitored fisheries. Ecuador fisheries were confined to the MCP off the port of Santa Rosa, Ecuador. The port of Anconcito is very near to Santa Rosa and thus not visible. Observed sets of the Peru driftnet shark fishery extended approximately to the 200m bathymetric contour in the waters fronting Salaverry port. Locations of seabird bycatch are represented by the black circles. Seabird bycatch species composition for each

fishery is indicated in a pie chart. Fishery abbreviations indicate gear type (longline [LL], or net) and main target species.

Table S1. Fisheries characteristics and observed effort summary for each of the fisheries monitored during the study, 2005 to 2011.

Table S2. Summary of seabird bycatch by country and fishery. Fishery abbreviation “LL” refers to longline fisheries. Capture species IUCN conservation status is listed in parenthesis: CR = critically endangered, EN = endangered, NT = near threatened, VU = vulnerable, LC = least concern.

Table S3. Seabird catch per unit effort (CPUE) comparison table. Bycatch categories include total catch for each fishery observed and for particular species or species groups. Species codes are: waved albatross (WVAL), black-browed albatross (BBAL), white-chinned petrel (WCPE). Fisheries are described by their gear type (net or longline) and by their main target species. CPUE estimates are averages of pooled monthly seabird bycatch during the study period, 2005 to 2011.

Figure S1. Mean pooled monthly bycatch of select seabird species in the (a) Ecuadorian demersal longline fishery for hake, (b) Ecuadorian surface longline fishery for yellowfin tuna, (c) Peruvian surface longline fishery for sharks and rays, (d) Peruvian surface driftnet fishery for sharks and rays. Seabird bycatch per unit effort is presented as catch per 1000 hooks for longline fisheries and catch per km^2h for the driftnet fishery. Seasonal fisheries (b) and (c) have their out-of-season periods indicated by the cross-hatched polygons.

Author's declaration of contribution to co-authored chapters/research papers

Chapter I: Post-capture movements of loggerhead turtles in the southeastern Pacific Ocean assessed by satellite tracking

Jeffrey C. MANGEL, Joanna ALFARO-SHIGUETO, Matthew J. WITT, Peter H. DUTTON, Jeffrey A. SEMINOFF and Brendan J. GODLEY

In this chapter I provide analyses of the movements of loggerhead turtles in relation to environmental and other variables after their incidental capture and release by small-scale longline fishing vessels in Peru. I wrote this chapter under the supervision of Dr. B. Godley. J. Alfaro-Shigueto assisted with project implementation, coordination and deployment of satellite transmitters on the turtles. Dr. M. Witt provided assistance with transmitter data management and spatial analyses. Dr. P. Dutton and Dr. J. Seminoff assisted with coordination of satellite transmitter deployments. This chapter was published in Marine Ecology Progress Series in 2011.

Chapter II: Latitudinal variation in diet and patterns of human interactions in the marine otter

Jeffrey C. MANGEL, Tara WHITTY, Gonzalo MEDINA-VOGEL, Joanna ALFARO-SHIGUETO, Celia CÁCERES and Brendan J. GODLEY

In this chapter I present an analysis of a multi-year dataset I collected on marine otter prey and activity patterns and compare that with similar data from Chilean research sites. I wrote this chapter under the supervision of Dr. B. Godley. T. Whitty provided assistance with data management and analysis for Peru. Dr. G. Medina-Vogel is lead researcher for the Chilean field component and provided prey species comparison data. Dr. Medina-Vogel also assisted with data analysis. J. Alfaro-Shigueto was the Peru project co-lead and was instrumental in coordinating data collection, management and analysis. C. Cáceres was involved in data collection and management. This chapter was published in Marine Mammal Science in 2011.

Chapter III: Small cetacean captures in Peruvian artisanal fisheries: High despite protective legislation

Jeffrey C. MANGEL, Joanna ALFARO-SHIGUETO, Koen VAN WAEREBEEK, Celia CÁCERES, Stuart BEARHOP, Matthew J. WITT and Brendan J. GODLEY

In this chapter I quantify and characterize small cetacean bycatch by Peruvian small-scale driftnet and longline fisheries based upon observer data from one port. I wrote this chapter under the supervision of Dr. B. Godley. J. Alfaro-Shigueto assisted with project implementation, logistics, data collection and management. K. Van Waerebeek provided input regarding catch characteristics and long term fishery trends. C. Cáceres was involved in data collection and management. Dr. S. Bearhop assisted with analyses of temporal trends in small cetacean bycatch. Dr. M. Witt provided assistance with spatial analyses of the bycatch data. This chapter was published in Biological Conservation in 2010.

Chapter IV: Using pingers to reduce small cetacean bycatch in the small-scale driftnet fishery in Peru

Jeffrey C. MANGEL, Joanna ALFARO-SHIGUETO, Matthew J. WITT, Dave HODGSON and Brendan J. GODLEY

In this chapter I present the results of a study testing the effectiveness of acoustic alarms at reducing small cetacean bycatch in the Peruvian small-scale driftnet fishery. I wrote this chapter under the supervision of Dr. B. Godley. J. Alfaro-Shigueto assisted with project implementation, logistics, data collection and management. Dr. M. Witt provided assistance with spatial and statistical tests of bycatch and pinger effectiveness. Dr. D. Hodgson assisted with statistical modeling of the impact of acoustic alarms on small cetacean bycatch. This chapter is in review with Oryx.

Chapter V: Onboard observer data suggest that small-scale fisheries are a major potential threat to seabirds in the southeastern Pacific

Jeffrey C. MANGEL, Joanna ALFARO-SHIGUETO, Andres BAQUERO, Jodie DARQUEA, Jessica HARDESTY NORRIS, Dave HODGSON and Brendan J. GODLEY

This chapter assesses and compares the seabird bycatch rates and patterns of multiple small scale net and longline fisheries in Peru and Ecuador. I wrote this chapter under the supervision of Dr. B. Godley. J. Alfaro-Shigueto assisted with project implementation, logistics, data collection and management. A.

Baquero and J. Darquea were responsible for data collection and management for the Ecuador fisheries assessed in the study. Dr. J. Hardesty Norris assisted with project coordination in Peru and Ecuador and assisted with the design of data collection protocols in Ecuador. Dr. D. Hodgson provided assistance with statistical analyses of seabird bycatch rates. This chapter is in preparation for submission to Biological Conservation.

Introduction

This thesis presents the results of research examining the interactions of Peruvian small-scale fisheries with threatened marine vertebrates. The subject matter and approaches of the chapters are diverse, reflecting the range of fishing methods employed, species present and conservation issues facing small-scale fisheries today. The chapters reflect efforts to better understand, characterize, and quantify fishery interactions, to examine the results of these interactions and to test technologies designed to mitigate these interactions.

In chapter one I provide an analysis of satellite tracking data of loggerhead turtles (*Caretta caretta*) captured incidentally by small-scale longline fishing vessels in southern and central Peru, and subsequently released (Mangel et al. 2011a). The purpose of the study was to assess the at-sea movements and distribution of loggerhead turtles in the region, determine if there were any apparent patterns of habitat selection, and to gain insights into post-release mortality and fisheries overlap. The work integrated tracking data on 14 juvenile and subadult loggerheads with data on sea surface temperature, bathymetry, political boundaries and regional fishing effort. The study was the first work of its kind with sea turtles in the southeastern Pacific Ocean region and is therefore a valuable contribution and expansion to our understanding of the southern Pacific loggerhead population, juvenile loggerhead behavior and distribution, and the degree to which loggerhead turtles and small-scale fisheries overlap in the region.

Chapter two focuses on the endangered marine otter (*Lontra felina*) and its prey selection and human interactions (Mangel et al. 2011b). In this study the main objectives were to examine the degree to which marine otters may be at risk from interspecific interactions, including anthropogenic, due to their use of artificial breakwaters at small-scale fishing ports as dens and shelters, and to assess marine otter diet for any latitudinal patterns. This work was based upon over 1,000 hours of focal and scan sampling of otters at two primary study sites in Peru and otter spraint sampling from a total of nine sites. The vast majority of studies of the marine otter have occurred in Chile (e.g. Medina-Vogel et al. 2006; Medina-Vogel et al. 2007; Medina-Vogel et al. 2004). This study is therefore important as a comparison study of the species nearing the northern limit of its range. The value of such work is highlighted by the novel finding of a latitudinal gradient in marine otter diet. This work also further documents the precarious state of the species due to coastal development and inter-specific interactions and details how proper management of artificial den sites could serve to promote species conservation.

In chapter three I present the results of a multi-year study to monitor the bycatch of small cetaceans by Peruvian small-scale drift gillnet and longline vessels (Mangel et al. 2010). The objectives of the study were to quantify and characterize small cetacean bycatch toward assessing the effectiveness of the ban on their capture and commercialization. Data were gathered through the use of onboard and shore-based observers who documented small cetacean bycatch and fishing activity. This is the first study in Peru to quantify small cetacean bycatch through the use of onboard observers and thus provides a more accurate assessment of bycatch rates than previous shore-based studies (e.g. Majluf et al. 2002; Read et al. 1988; Van Waerebeek and Reyes 1994b). Results of the study indicate that small cetacean bycatch may be occurring at a rate similar to before small cetacean exploitation was banned, and this is one of the highest reported small cetacean capture rates globally. The study also shows that small-scale fisheries can be effectively monitored and indicates that these fisheries can have impacts of a magnitude similar to commercial or industrialized fleets.

Chapter four builds upon the findings of chapter three and presents the results of a study testing the use of acoustic alarms (pingers) to reduce the bycatch of small cetaceans in the Peruvian small-scale driftnet fishery. Using onboard observers and quantifying small cetacean and target species catch rates on fishing sets that used (experimental sets) and did not use pingers (control sets) we were able to show that pingers reduced small cetacean bycatch by approximately 37%. Moreover, results indicate that fishing sets that used pingers experienced no change in the target species catch rate. While pingers have been shown to be effective in other fisheries (e.g. Barlow and Cameron 2003), this is one of the first study to implement the technology and show its effectiveness in a small-scale fishery in a developing nation. Given that small cetacean bycatch mortality rates in Peru likely exceed 10,000 animals per year, the implementation of this technology in this fishery could have sizeable impacts on small cetacean populations.

Chapter five focuses on the bycatch of seabirds by small-scale longline and gillnet fisheries in Peru and Ecuador. The objective of this study was to provide the first onboard observer based estimates of seabird bycatch for these fisheries for the region. The study placed special emphasis on monitoring the threat posed by these fisheries to the critically endangered waved albatross (*Phoebastria irrorata*). The work is primarily based upon onboard observer monitoring of seabird bycatch of nine small-scale fishing fleets in Peru and Ecuador. We observed bycatch of at least 16 seabird species including numerous endangered and threatened species. Bycatch of waved albatrosses in Ecuador was documented for the first time and, given the bycatch rates observed, could help in explaining the declines in adult survival observed at their Galapagos nesting colonies (Anderson et al. 2008; Awkerman et al. 2006). This work also represents one of the first attempts to characterize and quantify seabird bycatch in small-scale and coastal fisheries and suggests that they may have significant population impacts of some species and therefore warrant further study and work toward identifying effective bycatch mitigation measures.

Chapter I: Post-capture movements of loggerhead turtles in the southeastern Pacific Ocean assessed by satellite tracking

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Published in Marine Ecology Progress Series (2011) Volume 433: 261-272

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Abstract

This work offers insights into the post capture movements made by loggerhead turtles in the southeastern Pacific Ocean. From 2003-2007, 14 loggerhead turtles (*Caretta caretta*) were fitted with satellite transmitters and released off the coast of Peru. All turtles were juveniles (CCL range 40.5 to 68.5 cm) incidentally captured by small-scale longline fishing vessels from southern or central Peru. Tracking durations were highly variable (mean: 143 ± 90 d; range: 8 to 297 d) with no clear signs of immediate post-release mortality. Upon release, all turtles moved offshore beyond the continental shelf. Eight of 11 turtles tracked for >60 d had final displacements of less than 750 km suggesting that loggerhead turtles often maintain extended residency in these waters and that this area is an important foraging zone for loggerhead turtles of southwest Pacific origin. Satellite tracks also showed a substantial overlap of areas used by turtles with known Peruvian longline fishing effort. Turtles spent 75% of their time within the area fished by this fleet (based upon observed sets). This suggests that turtles are vulnerable to fishery interactions and that bycatch mitigation measures should be employed to minimize fishery impacts on loggerhead turtles. The loggerheads tracked during this study spent ca. 51% of their time in Peruvian waters, 39% on the high seas, and 9% in Chilean waters, emphasizing the need for a multinational approach to sea turtle conservation and fisheries management in the region.

Introduction

Many large marine vertebrates such as sharks, sea turtles, seabirds and marine mammals have complex life histories that encompass wide spatiotemporal scales. This situation places them at repeated risk of interaction with multiple anthropogenic threats which have been associated with the declines seen in many species (e.g. Baum et al. 2003; Read et al. 2006; Tasker et al. 2000; Wallace et al. 2010). Key among these threats is incidental capture (bycatch) in marine fisheries gear (e.g. longlines and driftnets) operating in pelagic zones (Hall et al. 2000). The loggerhead sea turtle (*Caretta caretta*), which is found worldwide in temperate and tropical waters (Bolten 2003), is one such species impacted by bycatch and is in decline in many parts of its range (Lewison et al. 2004b; Wallace et al. 2008).

The life history of loggerhead sea turtles operates at oceanic scales, with nesting sites, juvenile developmental habitats and adult foraging grounds for the same population often separated by thousands of kilometers (Bolten 2003). In the North Pacific Ocean, loggerhead nesting, which is confined to Japan, has been linked through genetic analyses and satellite tracking to aggregations of juveniles and subadults found in the central and eastern Pacific (Bowen 1995; Howell et al. 2010; Nichols et al. 2000; Polovina et al. 2006). In the South Pacific Ocean, decades of intense monitoring in eastern Australia has helped in defining many aspects of the nesting and oceanic ecology of loggerhead turtles in the western south Pacific and in describing declining trends in abundance for that population (Limpus and Limpus 2003a). However, it was not until 2004 that the species was confirmed in the southeastern Pacific Ocean (SEP) through a combination of onboard and shore-based fisheries monitoring (Alfaro-Shigueto et al. 2004). Genetic studies have shown that the juvenile loggerheads found in the SEP originate from Australian and New Caledonian rookeries (Alfaro-Shigueto et al. 2004; Boyle et al. 2009). Additional reports now also confirm the occurrence of loggerheads off Ecuador and Columbia (Alava 2008) and Chile to a southerly latitude of 32° (Donoso and Dutton 2010). Alfaro-Shigueto et al. (2008) report primarily small to medium sized juvenile and subadult sized animals found off the Peru coast (CCL range: 35.9-86.3cm) while Donoso & Dutton (2010) and Alava (2008) reported similar, if somewhat larger, sized loggerheads captured by industrial swordfish longline vessels in Chile (CCL range: 47-84cm) and small-scale longline vessels in Ecuador (CCL range: 50-80cm). It is noteworthy, however, that each of these studies was fishery dependent and thus may only represent those sizes of turtles vulnerable to capture by these particular fishing gears.

The wide dispersal of loggerhead turtles combined with their omnivorous feeding habits (which results in opportunistic foraging on fishing bait), leads to high levels of bycatch in coastal and pelagic fisheries, particularly near-surface longlines (Lewison et al. 2004b; Peckham et al. 2008; Tomás et al. 2008; Wallace et al. 2008). These fishery interactions have led to sizeable takes of loggerheads globally (Gilman et al. 2006; Lewison et al. 2004b; Wallace et al. 2008). As a result of this situation, the needs to improve our understanding of sea turtle pelagic life stages and interactions with fisheries have been identified as research priorities (Hamann et al. 2010).

There has also been growing awareness of the need to assess and address the issue of post-release mortality, that is, the potential for animals to die from fishery related injury after being released from fishing gear (Bjorndal et al. 2003; Hays et al. 2003). Such information is necessary in determining fishery related morality rates but is particularly challenging to evaluate without a means to track animals after their release. Assessments of post-release mortality based upon satellite tracking of released animals has been attempted with sea turtles, including loggerheads (Chaloupka et al. 2004; Howell et al. 2010; Sasso and Epperly 2007; Snoddy and Southwood Williard 2010; Swimmer et al. 2006) as well as with other marine megafauna including blue sharks (*Prionace glauca*) (Moyes et al. 2006) and blue marlin (*Makaira nigricans*) (Graves et al. 2002).

The SEP (Fig. 1) is one of four eastern boundary current systems on earth and is a highly dynamic and productive marine ecosystem (Carr 2002). It is dominated by the Humboldt Current System (HCS) which is a wind-induced coastal upwelling system. Winds in this system move along the Peruvian coast toward the equator before turning westward into the Pacific Ocean and result in year-round, nutrient rich upwelling (Bakun and Weeks 2008). This cold, nutrient rich coastal upwelling yields high levels of primary and secondary productivity and makes the HCS one of the world's most productive marine ecosystems (Carr 2002; Taylor et al. 2008).

Aside from supporting a considerable abundance of flora and fauna, the HCS also supports large fishing industries (Bertrand et al. 2004; Jahncke et al. 2004; Thiel et al. 2007). Indeed, approximately 16 - 20% of global annual fish production is derived from the region (Heileman et al. 2009). There are also very large and dispersed small-scale fleets, with approximately 40 000 small-scale vessels fishing the coastal and offshore waters of Ecuador, Peru and Chile (Alfaro-Shigueto et al. 2010; Arriaga and Martinez 2003b; OECD 2009). Bycatch of seabirds, cetaceans and sea turtles, including loggerhead turtles, has been documented in these small-scale fisheries in Peru (Alfaro-Shigueto et al. 2010; Awkerman et al. 2006; Gilman et al. 2010; Mangel et al. 2010). This is of concern given the 86% decline in loggerhead nesting reported for eastern Australia nesting beaches over 23 years despite local conservation efforts including nesting beach and in-water protected areas, hatchling management, and controls on some fisheries known to impact the species (Limpus and Limpus 2003b).

The purpose of this study was to improve our understanding of the ecology of loggerhead turtles in the SEP region. More specifically, we were interested in (1) assessing loggerhead at-sea movements and distribution, (2) determining habitat selection and (3) gaining preliminary insights into the effects of injury and potential for post-hooking mortality.

Materials and Methods

Turtle characteristics

Fourteen loggerhead turtles were fitted with satellite transmitters between March 2003 and February 2007 (Table 1). The turtles were incidentally entangled or hooked by longline gear set from small-scale fishing vessels targeting sharks in oceanic waters (mainly blue sharks and short-fin mako sharks, *Isurus oxyrinchus*) and dolphin fish (*Coryphaena hippurus*) and operating out of the ports of Ilo (71.33°W, 17.63°S) and Pucusana (76.78°W, 12.48°S) in southern and central Peru, respectively (Fig. 1). These vessels used Mustad classic J type fishing hooks with a 10° offset ranging in size from 1 to 3 (for a detailed description of the fishery see: Alfaro-Shigueto et al. 2010). The precise location of capture is known for 11 of the 14 animals. The study animals ranged in size from 40.5 to 68.5 cm curved carapace length (CCL, measured from nuchal notch to posterior most tip) and were thus classified as juveniles, as reported by Alfaro-Shigueto et al. (2008).

The turtles used in our study were captured by collaborating fishers during the final set of each fishing trip to allow prompt return to port for transmitter attachment. To further minimize time in captivity, animals were released within approximately 30 min of the completion of transmitter attachment several kilometers offshore, typically within 24 h of their original capture. All loggerheads used in the study were active at time of capture (not moribund or comatose) and fishers were given detailed instruction on how to safely handle and maintain the turtles aboard. All visible fishing hooks and entangling line were removed from the animals prior to release.

We assigned an injury score to each turtle based upon the location and severity of any visible injuries sustained during the capture process following criteria described by Chaloupka et al. (2004). We used a three point injury scale in which level 1 referred to animals with external injuries only (including those that were only entangled in longline branchlines), level 2 indicated minor injuries to the mouth cavity or lower mandible, and level 3 indicated more severe injuries including animals deeply hooked in the esophagus or soft palate. This examination was primarily external in nature (except in those cases of level 3 injuries) and was not meant as an overall turtle health assessment (as in Heithaus et al. 2007), but rather, as a scoring of the injuries associated with the animal's capture.

Transmitter application and data analysis

We used Telonics (Mesa, Arizona, USA) and Wildlife Computers (Redmond, Washington, USA) satellite transmitters (Platform Transmitter Terminals, PTTs) in the study (Table 1). We attached PTTs to the anterior central scutes of the carapace using the fiberglass cloth and polyester resin method described by Balazs et al. (1996) or using PowerFast™ two-part marine epoxy (Coyne et al. 2009; Table 1). PTTs had one of two laddered duty cycles (Table 1) where Duty cycle 1 was a set to transmit 24 h on (month 1), 6 h on / 13 h off (months 2 to 3), and 6 h on / 25 h off (>month 3) and Duty cycle 2 was set to transmit 24 h on (month 1), 24 h on / 48 h off (month 2 to 3), 24 h on/72 h off (month 4 to 5), and 24 h on/96 h off (>month 6).

Positional data were received from Service ARGOS and managed using the Satellite Tracking and Analysis Tool (STAT; Coyne and Godley 2005). Argos positional data are accompanied by indicators of their spatial accuracy where positions assigned location class 3 are of greatest accuracy (<350m), and those with location class 0 are of least accuracy (>1km). Positions assigned classes A and B have no estimate of their accuracy. These data were subjected to a combination of filtering procedures to eliminate potential outliers. ARGOS derived location classes 3,2,1,0,A and B for turtle positions were retained for this analysis (Witt et al. 2010) and filtered based upon speed (>5km h^{-1} excluded; Luschi et al. 1998) and turning angle (<25° excluded). Tracks were also reduced to one location per day in order to produce an unbiased data set of all environmental and behavioral variables (See De Solla et al. 1999 for further discussion of handling autocorrelation in animal movement datasets). For days when more than one location was obtained, the position of highest quality ARGOS location class was retained. If multiple positions for a given day were of the same highest quality, the location nearest to 12:00 pm local time was selected. Following track filtering, the first seven days of each track after release were eliminated to minimize any artifact introduced by transporting and releasing animals relatively close to the coast. We used ESRI ArcMap 9.2 (Redlands, California, USA) to display turtle positions and the Hawth's Tools Extension (www.spatialecology.com/htools) to create minimum convex polygons of longline fishing effort.

Longline fishing effort data detailed in Alfaro Shigueto et al. (2008) was utilized in this study to examine the overlap with the turtles tracked in this study. Trained onboard observers collected data on fishing effort and catch, including set positions (using handheld GPS), aboard fishing vessels from eight ports (242 trips, 1773 sets) for the years 2004 to 2007. This is the same fleet from which the loggerhead turtles in this study were captured. A polygon of Peruvian fishing effort was created by combining minimum convex polygons of fishing effort from the eight ports.

Satellite tracking data were compared with bathymetry and sea surface temperature data. Bathymetric values were obtained from the Global Bathymetric Chart of the Oceans (GEBCO, www.gebco.net, IOC et al. 2003). An eight day composite 4 km spatial resolution Nighttime (4 micron) Sea Surface Temperature Dataset from the MODIS-Aqua satellite was used to determine SST at the location of each turtle's daily position (Feldman & McClain 2009). Maritime political boundaries were obtained from the Flanders Marine Institute Maritime Boundaries Geodatabase Version 5.0 (http://www.vliz.be). Time within a given boundary was calculated by tabulating entry and exit dates and times from the tracking data and assigning those time periods to their appropriate polygons. Any period in excess of seven days which could not be accounted for due to lack of positions was discarded (2% of cases) but times less than seven days were attributed to the dominant polygon (7% of cases). Data regarding environmental variables, as well as speed of movement, displacement from release location (including seven day release period) and political boundaries were limited to animals tracked for 60 d or more (n = 11). Swim speeds are reported as 'minimum overall average' since our speed calculations assume straight line travel between consecutive positions. Descriptive statistics are presented as mean ± standard deviation.

Results

Environmental variables

Following their release, all animals returned to oceanic waters (Fig. 1 and Supplemental Fig. 1) and did not appear to return to coastal waters (< 200 m). The overall mean depth of the waters occupied was 4286 ± 376 m (range of means: 3392 to 4704 m, median = 4352 m, n = 11). The average SST was 21.1 ± 2.2°C (range of means: 16.2°C - 23.8°C, median = 21.6°C, n = 11; Fig. 2), with only 4% of SST values less than 15°C.

Track durations

Tracking durations ranged from 8 to 297 d (mean duration: 143 ± 90 d, median = 129 d, n = 14; Table 1). The minimum overall average swim speed was 0.70 ± 0.11 km h^{-1} (range = 0.57 km h^{-1} to 0.90 km h^{-1}, median = 0.66 km h^{-1}, n = 11). There was no effect of animal size on track duration (randomization test with 10,000 iterations and within and between group randomization, two-tailed p = 0.96; Fig. 3a). Likewise, there was no effect of level of injury upon track duration (Randomization test, two-tailed p = 0.99; Fig 3b). There was also no effect of either animal size (Randomization test, two-tailed p = 0.96; Fig 3c) or injury score on minimum overall average swim speed (Randomization test, two-tailed p = 0.12; Fig 3d).

The number of uplinks received per day for each turtle was also examined because an increase in daily signals received could indicate that an animal is floating injured or dead at the surface (Hays et al. 2007). There was no evidence of an increase in daily signals received for any of the tracked animals (Supplemental Fig. 2). We also reviewed the battery voltage information for the four SPOT5 tags (the only tags for which this information was available) to assess transmitter battery life as a possible reason for termination of the tracks for those animals. There was no sign of voltage declines sufficient to halt transmission toward the end of the track periods for any of the four tags.

Displacement

Maximum displacement ranged from 372 to 2337 km (mean: 921 ± 667 km, median = 593 km; Table 1, Fig. 1, Supplemental Fig. 4). Eight of the eleven turtles with tracks of greater than 60 d (track duration range 79 to 223 d) had a final displacement of less than 750 km. The remaining three animals had track durations ranging from 249 to 297 d and maximum displacements from 1607 to 2337 km.

There was no effect of animal size on displacement rate (randomization test, two-tailed $p = 0.64$; Fig. 3e), nor was there an effect of capture to release distance on maximum displacement ($r = 0.07$, Supplemental Fig. 3a). However, there was an effect of level of injury upon displacement rate (Randomization test, two-tailed $p = 0.002$; Fig 3f, Supplemental Fig 4) with animals with injury scores 2 and 3 having greater displacement rates than animals with injury score 1.

Logistical constraints meant that animals could neither be fitted with transmitters *in situ* nor returned to capture location for release. The average distance between capture and release locations was 261 ± 153 km, (range: 98 to 593 km, median = 245 km, n = 11) but there was no correlation between capture to release distance and distance traveled after one month from the release date ($r = 0.02$, $p = 0.96$, Supplemental Fig. 3b). There was no clear evidence of high precision homing *per se* although animals approached to within 119 ± 75 km (range of minimum distances: 25 to 248 km, median = 101 km, n = 11) of their capture location and did so 34 ± 33 d (range: 3 to 85 d, median = 18 d) post-release (Table 1). As would be expected, there was a correlation between capture to release distance and nearest approach to capture location ($r = 0.66$, $p < 0.05$, Supplemental Fig. 3c).

Fisheries and Governance

Although captured and released in Peruvian waters, our study animals also moved within Chilean and International waters during tracking periods (Peru: mean 51% ± 29%, median = 59%, range for individuals: 7 to 97%; Chile: 10% ± 24%, median = 1%, range: 0 to 82%; High Seas: 39% ± 30%, median = 32%, range: 0 to 93%; Table 1, Fig. 1). The combined tracks of all turtles covered an area of approximately 2500 km from east to west and of 1600 km from north to south. There was also a large overlap of turtle movements with longline fishing areas. Animals spent 75% ± 33% (range: 13% to 100%, median = 99%) of their time within previously defined Peruvian small-scale longline fishing grounds

(Alfaro-Shigueto et al. 2008). There was no effect of animal size on time spent within the fishing grounds (Randomization test, two-tailed $p = 0.86$; Fig 3g).

Discussion

This work offers insights into post-capture movements of loggerhead turtles in the SEP and represents the first work of its kind with sea turtles in the region. As such, while based upon a relatively small sample, it provides a valuable point of comparison with similar studies of conspecifics elsewhere and can help inform regional and global efforts to better understand fishery impacts (Alessandro and Antonello 2010; Blumenthal et al. 2006; Lewison et al. 2004b; McClellan et al. 2009; Wallace et al. 2008) and post-release mortality (Bjorndal et al. 2003; Chaloupka et al. 2004; Hays et al. 2003; Sasso and Epperly 2007; Swimmer et al. 2006).

Environmental factors

Our work indicates that many of these animals were "resident" in the waters off Peru and Chile where they maintained a pelagic lifestyle for the duration of tracking. Animals spent greater than 97% of their time in waters in excess of 1000 m depth. Moreover, eight of 11 turtles had final displacements of less than 750 km from release even though tracking durations extended up to 223 days. These results, while based upon instrumented animals released at locations distinct from their site of capture, support findings from other regions indicating that juvenile loggerheads may be actively selecting key pelagic habitats and are not simply passively distributed by ocean currents (Hays et al. 2010; McCarthy et al. 2010; Monzón-Argüello et al. 2009; Polovina et al. 2006). Furthermore, given the sizes of turtles in this study, this would also be consistent with a transition by juvenile loggerheads from passive to active swimmers at approximately 40 to 60 cm SCL, as has been reported in the Atlantic and Mediterranean (Bolten 2003; Cardona et al. 2005; Revelles et al. 2007). However loggerheads CC4, CC5 and CC14 did make relatively long movements both north and west which match the general surface current patterns in the region (Fig. 1). Therefore, loggerhead movements may comprise a combination of active station holding and passive current driven drifting (Cardona et al. 2005; Hays et al. 2010).

Studies off the Baja California Peninsula indicate that juvenile loggerhead turtles may take up residency for extended periods before returning to the western Pacific (Etnoyer et al. 2006; Nichols et al. 2000; Peckham and Nichols 2003; Seminoff et al. 2004). Our findings suggest that a similar scenario occurs in the waters off Peru and northern Chile. Moreover, as has been observed in several studies of juvenile loggerheads in the Mediterranean, the animals in our study appeared to avoid the waters of the continental shelf (Cardona et al. 2005; Monzón-Argüello et al. 2009; Revelles et al. 2007). This is reinforced by similar findings by Donoso & Dutton (2010) who report loggerheads in pelagic waters offshore from northern Chile. Given the size class of animals in the current study and for the region (Alfaro-Shigueto et al. 2008), which is primarily the oceanic juvenile stage (Bolten 2003; Limpus and Limpus 2003a), this may be as expected. The findings of Howell et al. (2010) that juvenile loggerheads in the central north Pacific spent >90% of their time in the upper 15 m of the water column supports the notion of a pelagic foraging lifestyle for this life history stage in loggerheads. Furthermore, if these juvenile animals have poorly developed diving abilities (Bolten 2003; Cardona et al. 2005; Revelles et al. 2007) then they would not be able to exploit benthic prey found over the shelf and may therefore specialize in pelagic prey, similar to Baja California where the loggerhead population has been shown to feed primarily or exclusively upon pelagic red crab (*Pleuroncodes planipes*) (Nichols et al. 2000; Peckham and Nichols 2003).

The mean SST of 21.1 ± 2.2°C experienced by these animals was well within their thermal tolerance (Coles and Musick 2000; Milton and Lutz 2003; Witherington and Ehrhart 1989), but the minimum SST experienced by some individuals reached below 15°C on several occasions and could therefore be approaching the species' lower thermal tolerance (Coles and Musick 2000; Milton and Lutz 2003). As Howell et al. (2010) note, however, this lower limit may be better thought of as a species preference and less as an absolute value. But these cool temperatures do suggest the potential for seasonal north-south movements, as seen in the western Atlantic (Coles and Musick 2000; Hawkes et al. 2007; Mansfield et al. 2009) and Mediterranean (Bentivegna et al. 2007) that, in cooler months, likely bring loggerheads that have been recorded in waters off northern Chile (Donoso and Dutton 2010) north into the relatively warmer waters of Peru. It is also interesting to note that Donoso & Dutton (2010) report an increase in loggerhead bycatch associated with an incursion of a 21°C warm zone for the year 2001, and also found a bimodal association of SST (18°C and 21°C) with the occurrence of loggerheads off Chile. This coincides closely with the mean SST for the loggerheads tracked in our study and with findings in the central north Pacific that loggerhead bycatch by the Hawaii based longline fleet was highest when setting at oceanic fronts of 17°C and 20°C (Polovina et al. 2000). That loggerheads may aggregate in waters of certain temperatures may also help explain their absence from continental shelf waters in Peru. These areas are dominated by cold coastal upwelling waters ranging from 15 to 17°C in the winter and 15 to 19°C in the summer months (Bertrand et al. 2004). The apparent avoidance of the continental shelf by loggerheads in our study may therefore be driven by one or more of the (possibly related) drivers of avoidance of cold coastal waters, active selection of preferred foraging habitat, or lower risk of predation (Bolten 2003; Eckert et al. 2008; Hawkes et al. 2007). Additional studies to further detail loggerhead prey species and foraging behavior in the SEP would help resolve this question.

Fisheries and Governance

The turtle movements we describe overlapped with the area operated by the Peruvian small-scale longline fishery (Fig. 1; Alfaro-Shigueto et al. 2008). Indeed, they fell within the fishing grounds of vessels monitored from eight ports for 75% of their track durations and half of the study animals spent the entire track period within the fishing zone. This profoundly underestimates the extent of overlap with this one fishery since it operates from at least 19 ports and sets an estimated 80 million hooks per annum (Alfaro-Shigueto et al. 2010). Alfaro-Shigueto et al. (2010) also note that the Peruvian small-scale longline fleet sets their mainline at the ocean surface. Therefore, if juvenile loggerheads spend >90% of their time within the upper 15 m, as has been reported in the central north Pacific (Howell et al. 2010), then they have a heightened risk of interacting with this fishing gear. There are, in addition, other fisheries such as driftnets, industrial purse-seine, and high seas longline fleets that have not been assessed but are of concern because of their potential for interactions with loggerheads in the region.

Chaloupka & Limpus (1997, 2001) have reported on fisheries in the western Pacific that have bycatch of loggerheads. But it is now clear that at least some of the loggerheads from the Australian and New Caledonian stocks spend extended periods of time outside the western Pacific. Research has shown that loggerhead maturation, and the oceanic movements that accompany it, may take decades, with age at maturity estimates ranging from 10 to 30+ years (Bjorndal and Bolten 2001; Bjorndal et al. 2000; Casale et al. 2009; Chaloupka 1998; Parham and Zug 1997; Zug et al. 1995). This implies extended periods, possibly decades, in a given life stage and its accompanying habitat during which pelagic juveniles in particular are exposed to contact with the other fisheries operating on the high seas. While there is some information for the central north pacific for loggerhead interactions with the Hawaii based longline fleet, similar information is absent for the central south Pacific. However, Domingo et al. (2010) observed a bycatch of loggerheads by Uruguayan flagged vessels operating in this region. Donoso & Dutton (2010), document loggerhead interactions with the Chilean industrial swordfish longline fishery, but also note the lack of information on other fleets in the area, including the Spanish longline fleet operating out of southern Peru. Peckham et al. (2007; 2008) have shown for the North Pacific that fisheries impacts on loggerheads, including small-scale (or artisanal) fisheries, can be severe. Likewise, in other regions and with other sea turtles species, similar, but not fully understood, trends of prolonged exposure to fishery interactions, including to small-scale fisheries, have been reported (Alfaro-Shigueto et al. 2007; Casale 2011; Lewison et al. 2004b; Wallace et al. 2010). There is clearly a need for a full, detailed assessment of bycatch and potential mitigation measures in the Peruvian small-scale fleet as well as the full suite of substantial industrial and small-scale fisheries operating in the region, especially given the rapid and sustained decline of this species in Australia (Limpus and Limpus 2003b).

Injury effects

One of the objectives of this study was to use satellite tracking movement data to gain insights into the effects of injury to bycaught animals. We found no impact of injury on track duration. However, there appears to have been an impact of injury on the rate of displacement. Animals we scored as having minor or severe injuries (scores 2 and 3) displaced at a much faster rate than animals with only external injuries. The reason for this difference is not clear. Animals with the fastest displacements tended to move in the same general direction as main surface currents in the region so could be exhibiting some passive drifting (Fig. 1 inset). But the long duration and characteristics of the tracks (i.e. battery voltage, uplinks day^{-1}) suggest that injuries were not fatal. We also acknowledge that the animals in this study may have had pre-existing injuries that were not visible or accounted for in our ranking. Thus our results, while indicative of the animals' injury status, do not represent a complete understanding. Similar to Howell et al. (2010) it is also noteworthy that two of our longest track durations came from animals we categorized as having severe injuries. Sasso & Epperly (2007), reporting on juvenile loggerheads in the North Atlantic, also observed that lightly hooked loggerheads in their study had a similar survival rate to uninjured, control animals. While acknowledging the small sample size of this study, these results may suggest that loggerheads are able to survive for extended periods with injuries, including severe injuries. Or they might indicate that our understanding of what entails a minor or severe injury to a sea turtle is incomplete. One transmitter failed after only 8 days but could not be attributed to death of the turtle, nor could the remaining turtles, which transmitted from 48 to 297 days. Use of PTTs or pop-up satellite archival tags (PSAT) with dive data or depth sensors could help make more informed determinations, but still suffer from a similar lack of clarity in revealing an animal's fate (Chaloupka et al. 2004; Swimmer et al. 2006). Future studies in this fishery should determine the prevalence of entanglements as well minor and severe hookings in order to help evaluate their likelihood of survival and the relative risk posed by the fishery. The impacts of sub-lethal injuries also need to be explored further as these could lead to reduced fitness as a result of tissue damage, infection, impaired feeding or swimming, or greater predation risk (Sasso and Epperly 2007; Watson et al. 2005), and may be particularly important in areas where turtle habitat strongly overlaps fishing grounds and turtles thus may face repeated capture and injury. Studies of longline fisheries bycatch of loggerheads in the region do indicate that the majority of bycaught loggerheads are released alive (Alfaro-Shigueto et al. 2008; Donoso and Dutton 2010), as do reports of longline fisheries in other regions (Casale 2011; Gilman et al. 2007; Kotas et al. 2004). This situation could be further improved through fisher training in sea turtle safe handling and release methods and the adoption of bycatch mitigation measures such as the use of circle hooks or mackerel type bait, which have been shown to reduce sea turtle bycatch and injury type and severity in other fisheries (Gilman et al. 2007; Watson et al. 2005; Yokota et al. 2009).

The pelagic distribution of loggerheads observed here highlights the challenges of research on this species in the SEP region, and of other highly mobile, pelagic species, generally. Monitoring through the use of onboard observers on fishing vessels has provided valuable information on size classes and distribution (Alfaro-Shigueto et al. 2004; Alfaro-Shigueto et al. 2008). But this type of fishery dependent data does not necessarily describe the full ecology of the species in the region. Here we have reported on the movements of loggerhead turtles bycaught by small-scale longline fishing vessels. While we have limited our analyses to only those animals that have tracks of 60 days or more (as a means to control for potentially aberrant behavior by injured animals) it is possible that these tracks do not fully represent the normal habitat or behavior of the species in the region. We also recognize that other variables could have an impact on track duration (i.e. attachment method, location of capture, transmitter type, etc.) but small sample size limits the degree to which these variables can be fully explored. But given the extremely high level of fishing effort in the region (Alfaro-Shigueto et al. 2010) and the resulting high likelihood of interactions between fisheries and loggerheads, this information is extremely valuable in improving our understanding of the species in the SEP and its interactions with fisheries.

Future directions

Much remains to be learned about loggerheads in the SEP. While we believe results from the present study provide extremely useful information on many aspects of loggerheads in the SEP, satellite tracking of uninjured animals is recommended to obtain additional fishery independent data. Moreover, similar to research in other ocean basins and with other species (e.g. Howell et al. 2010; Moyes et al. 2006; Swimmer et al. 2006), information on loggerhead dive profiles and possible relationships to oceanographic variables like currents, fronts and eddies would be extremely valuable for defining three dimensional habitat use and further assessing their vulnerability to fisheries bycatch. Such research can also help inform efforts to manage these fisheries so as to minimize opportunities for sea turtle bycatch (Blumenthal et al. 2006; Godley et al. 2010; McClellan et al. 2009).

As we have noted, recent work has helped to better characterize some Peruvian and Chilean fisheries and their sea turtle interactions (Alfaro-Shigueto et al. 2010; Alfaro-Shigueto et al. 2008; Donoso and Dutton 2010) but almost nothing is known about those interactions in the other small-scale and industrial fleets from many nations operating in the southern Pacific Ocean. Evidence of interactions with Uruguayan flagged commercial longliners operating in the Pacific Ocean have been reported (Domingo et al. 2010), but there remains a vast swath of the central south Pacific which loggerheads most likely inhabit for many years before they return to neritic habitats in the western Pacific. There is clearly a need for a more complete assessment of the fisheries loggerheads are likely to encounter during that time in order to better identify, categorize and rank the threats they face and to identify and implement effective mitigation strategies.

Acknowledgements

We warmly thank the participating fishermen, Captains Miguel Cuentas and Francisco Bernedo in particular , for their generous collaboration. Pro Delphinus biologists and veterinarians D. Montes, and M. Pajuelo provided superb field work. The artisanal fisher association from the port of Ilo (SUBCABPI) provided excellent logistical support. The manuscript benefitted from helpful comments from four anonymous reviewers. Project funding and equipment was provided by NOAA-NMFS Southwest Fisheries Science Center and the National Fish and Wildlife Foundation. JCM and JAS are Overseas Research and Students Awards Scheme (ORSAS) and University of Exeter scholarship awardees, respectively. JCM, JAS and BJG receive funding from Darwin Initiative. MJW is funded by the Peninsula Research Institute for Marine Renewable Energy (PRIMaRE). Research was authorized under Instituto Nacional de Recursos Naturales (INRENA) permits 015-2002-INRENA-J-DGFFS-DCB, 070-2003-INRENA-IFFS-DCB, 068-2004-INRENA-IFFS-DCB, 025-2005-INRENA-IFFS-DCB, and 002-2006-INRENA-IFFS-DCB.

Table 1. *Caretta caretta*. Summary of environmental and tracking variables for 14 loggerhead turtles captured as bycatch off the coast of Peru. Ports are Ilo (I) and Pucusana (P). PTT models SPOT5 and SDR-T16 were obtained from Wildlife Computers while the remainder (ST-18, ST-20, ST-20D, A1010) were from Telonics. Attachment methods referred to are using fiberglass resin (F) or Powerfast epoxy (P). Duty cycle 1 was a laddered program set to transmit 24 h on (month 1), 6 h on and 13 h off (months 2 to 3), and 6 h on and 25 h off (>month 3). Duty cycle 2 was a laddered program set to transmit 24 h on (month 1), 24 h on and 48 h off (month 2 to 3), 24 h on and 72 h off (month 4 to 5), and 24 h on and 96 h off (>month 6). Injury score 1: external injuries (including entanglement); injury score 2: minor internal injuries to mouth or lower mandible; injury score 3: severe injuries due to deep hooking. D refers to displacement (Columns 10 and 11). CCL = curved carapace length

Turtle	CCL (cm)	Deploy date	Port	PTT model	Attach method	Duty cycle	Injury score	Track duration (d)	Max D (km)	D rate (km d^{-1})	Maritime zones (%) Peru	Chile	High Seas	Time in fishing zone (%)	Capture to release distance (km)	Closest approach to capture (km) | (d)
CC1	63.2	3/25/03	I	ST-18	F	1	1	223	446	2.00	16	82	2	100	-	-
CC2	64.0	10/24/03	I	ST-18	F	1	1	136	479	3.52	73	0	27	39	394	248 | 71
CC3	68.0	10/24003	I	ST-18	F	1	1	142	593	4.18	78	7	15	55	399	89 | 85
CC4	61.5	4/7/05	I	ST-20	F	1	3	297	2,337	7.87	29	4	67	20	295	118 | 55
CC5	65.9	7/14/05	I	SDR-T16	F	1	3	289	1,658	5.74	7	0	93	13	-	-
CC6	56.5	2/3/06	P	ST-18	F	1	1	147	672	4.57	33	0	67	100	249	228 | 5
CC7	54.2	2/5/06	P	ST-18	F	1	1	114	581	5.10	59	0	41	100	229	142 | 8
CC8	68.5	11/14/06	P	ST-20D	F	1	1	94	522	5.55	68	0	32	98	593	187 | 81
CC9	65.5	1/13/07	I	SPOT5	F	2	1	79	372	4.71	66	11	23	100	124	62 | 18
CC10	64.1	1/13/07	I	SPOT5	P	2	2	51	395	7.74	0	63	37	100	117	48 | 4
CC11	40.5	1/13/07	I	SPOT5	P	2	2	48	361	7.53	16	75	9	100	126	58 | 34
CC12	60.8	1/16/07	I	SPOT5	P	2	3	8	60	7.50	100	0	0	100	-	-
CC13	51.3	2/5/07	I	A-1010	P	1	2	121	721	5.96	97	3	0	70	98	25 | 3
CC14	65.0	2/8/07	I	A-1010	P	1	2	249	1,752	7.04	35	1	64	61	245	101 | 7

Figure 1. *Caretta caretta*. Map of all turtle track locations (60+ day turtles) by level of injury and showing a polygon of longline fishing effort monitored from eight ports (242 trips, 1771 sets) collected by fisheries observers from 2000 to 2007 (Alfaro-Shigueto et al. 2008). Gray shading of tracks indicates injury scores: light gray = injury score 1 (n=7), dark gray = injury score 2 (n=4), black = injury score 3 (n=3). The termination points of tracks of injury scores 2 and 3 are also marked with shaded squares and triangles, respectively. Tracked loggerhead positions were within fishing area boundaries from 75% ± 33% of the time (Range 13% to 100%), (250 m, 750 m, 2000 m, and 3000 m bathymetric contours are also shown). Inset map shows the predominant current patterns (arrows) of the southeastern Pacific Ocean. MCP: minimum convex polygon

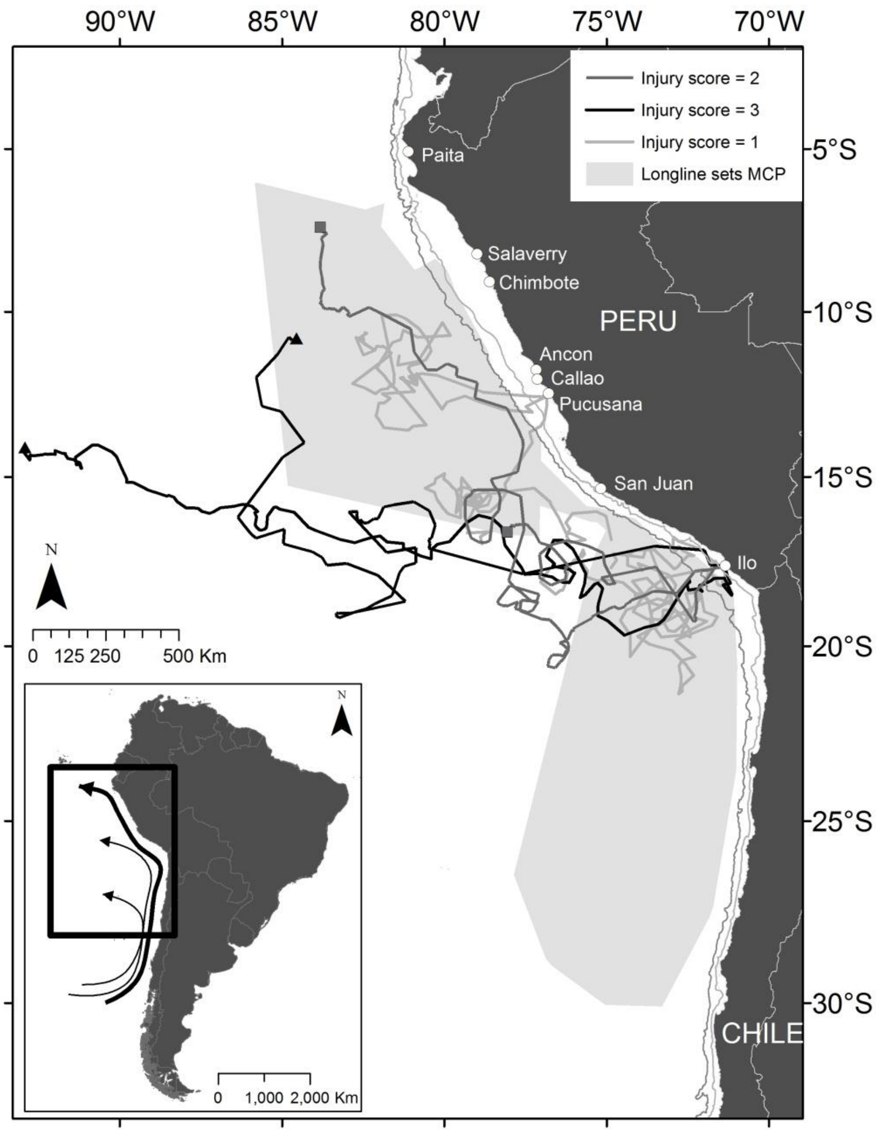


Figure 2. *Caretta caretta.* Pooled monthly SST for all turtles tracked for >60 d (n = 11).

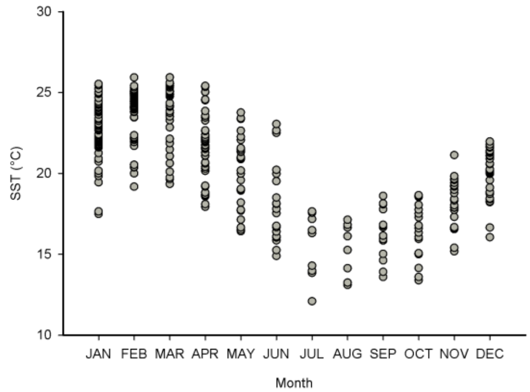
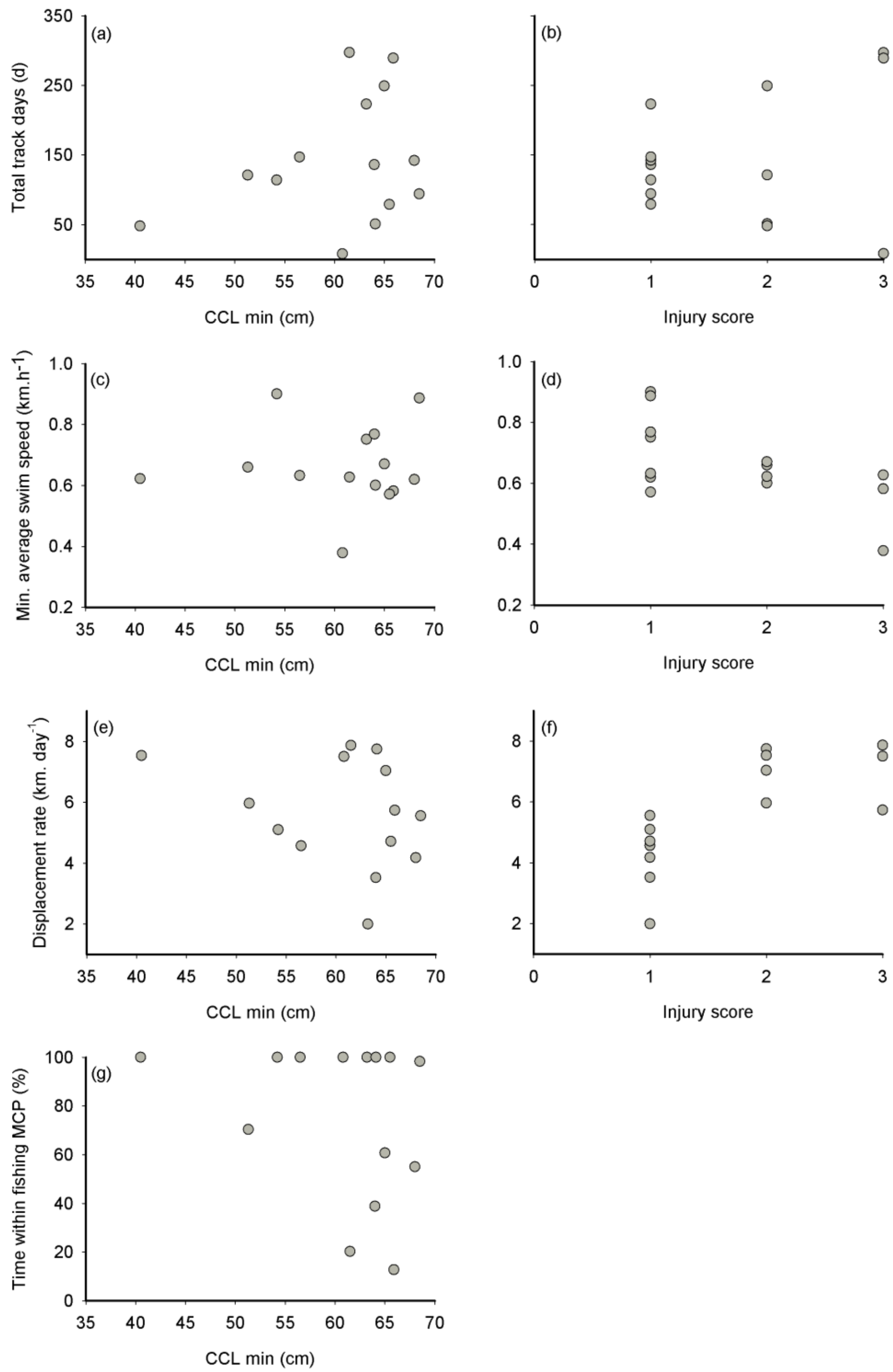


Figure 3. *Caretta caretta.* Track durations, minimum overall average swim speeds, displacement rates and time within fishing grounds minimum convex polygon (MCP) grouped by turtle size (curved carapace length, CCL) and injury score.



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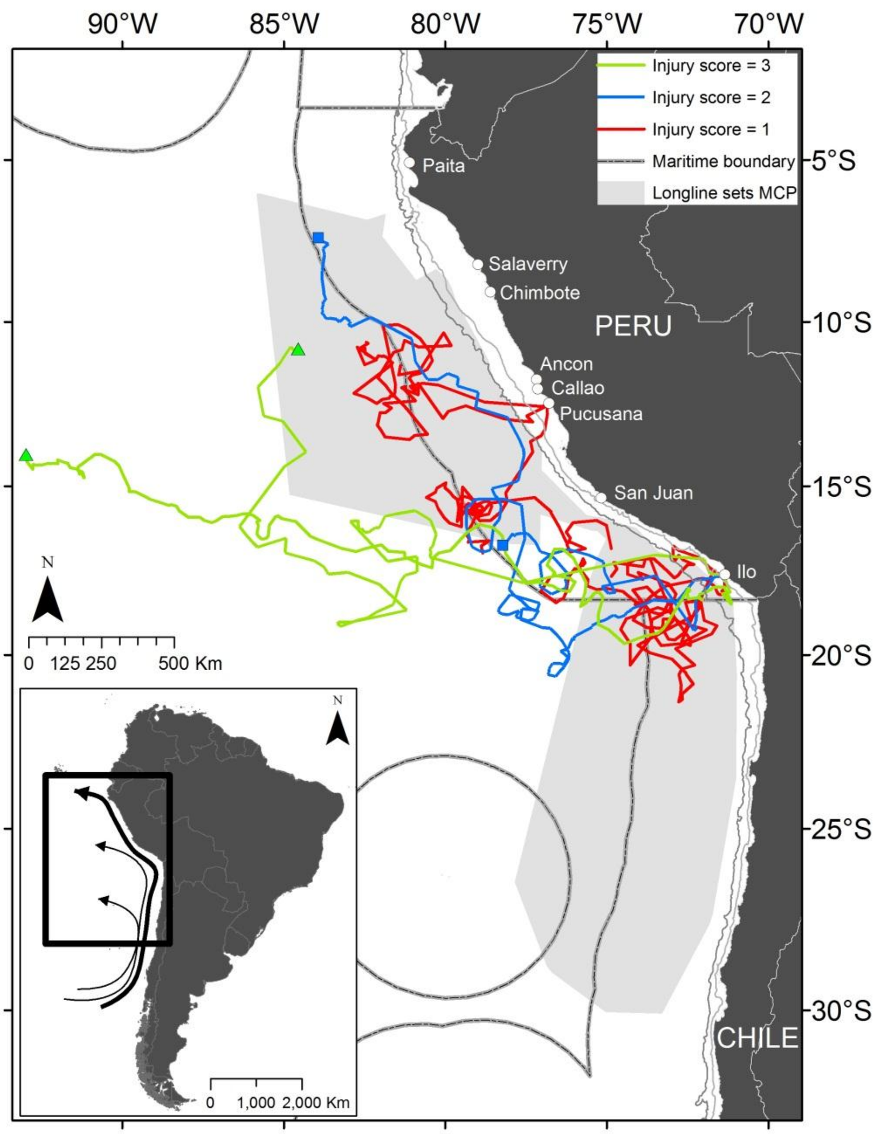


Figure S2. *Caretta caretta*. Number of uplinks received per day for each turtle for last 45 days of the track. X-axis is days until track termination beginning at day 45. There is no evidence for a spike in uplinks toward the end of any track which could indicate that a turtle was floating injured or dead at the ocean surface. A change in transmitter duty cycle can be clearly noted in tracks CC10 and CC11 at approximately the 30 d mark.

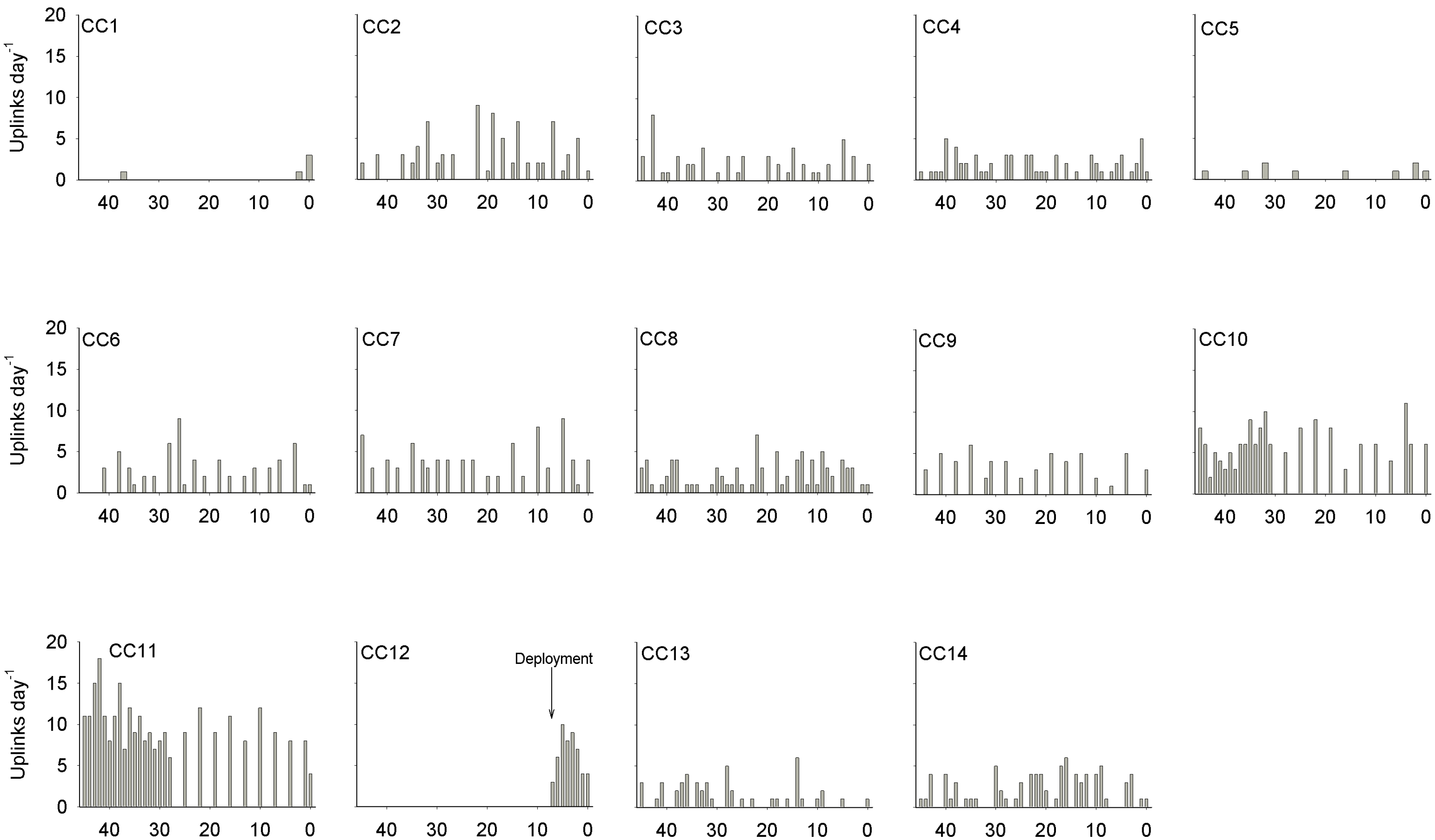


Figure S3. *Caretta caretta.* Scatterplots showing the relationship of distance of release location from capture location with (a) maximum displacement (r^2 = 0.0053; n = 11), (b) total distance traveled after 1 mo (r^2 = 0.0003; n = 10), and (c) closest approach to capture location (r^2 = 0.436; n = 11).

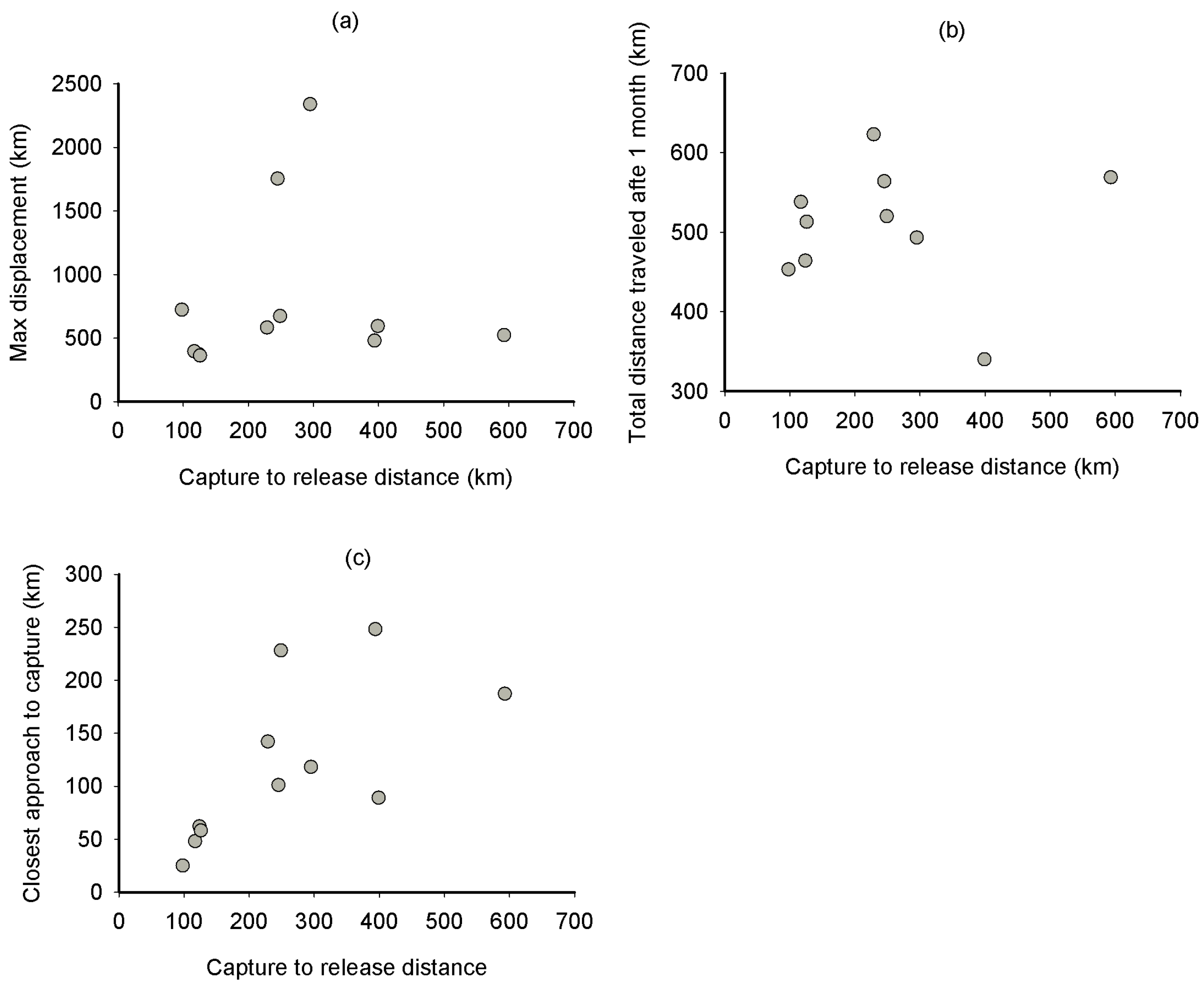
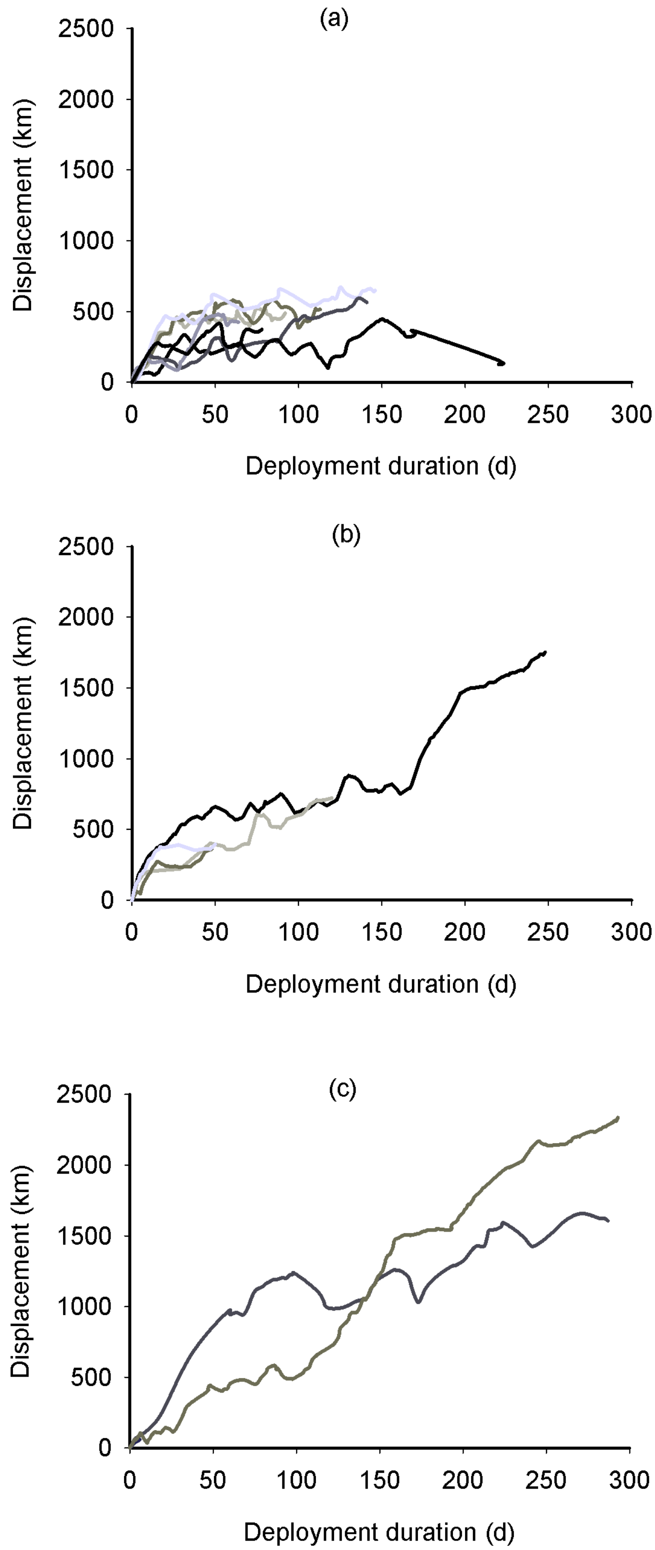


Figure S4. *Caretta caretta.* Daily displacements grouped by Injury scores 1, 2 and 3 (Panels a, b and c, respectively).



Chapter II: Latitudinal variation in diet and patterns of human interaction in the marine otter

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Published in Marine Mammal Science (2011) Volume 27(2): E14-E25

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Abstract

The marine otter (*Lontra felina*) inhabits patches of rocky coastline from central Peru to southern Chile and is classified as Endangered by the IUCN. Given the limited information available about the species, we set out to assess marine otter diet with a view to detecting latitudinal differences, and to assess marine otter activity budgets and inter-specific interactions (including anthropogenic) at Peruvian fishing villages and to compare results with similar Chilean studies. Nine study sites from central Chile to southern Peru were sampled for otter spraints to assess relative frequency of prey types and two fishing ports in southern Peru were monitored through focal and scan observations to assess activity patterns, inter-specific interactions, habitat use patterns, and dive durations. Results indicate that toward the northern part of its range, crustaceans become less important and fish more important in the diet. Interactions were observed between marine otters and other species, including stray dogs and cats. The strong dependence of marine otters on the availability of safe rocky shelters, and the species' apparent tolerance to living alongside humans raise conservation concerns about vulnerability to anthropogenic threats. These factors, if not correctly managed, could turn some of these rocky seashore patches into population sinks.

Introduction

The marine otter (*Lontra felina*) has a limited and patchy distribution that ranges from central Peru (6°S) to the southern tip of Chile (56°S) (Alvarez and Medina-Vogel 2008; Medina-Vogel et al. 2008), with isolated populations in Argentina (Larivière 1998). The species was historically hunted for pelts in Peru and Chile, resulting in considerably reduced abundance and geographic distribution (Brownell 1978; Cabello 1978; Iriarte et al. 1997; Iriarte and Jaksic 1986). The species population is thought to be declining and is classified as Endangered (IUCN 2008).

Published studies on marine otter distribution, activity budgets, prey composition and anthropogenic effects are limited to populations in Chile (Medina-Vogel et al. 2006; Medina-Vogel et al. 2007; Medina-Vogel et al. 2008; Medina-Vogel et al. 2004; Medina 1995a, b; Ostfeld et al. 1989). Similar studies of marine otters in Peru are confined to unpublished works. Previous research in Chile suggests that foraging is the dominant category of activity while marine otters are in sight, and that otters feed opportunistically, with diet primarily comprised of crabs, followed by fish (Medina-Vogel et al. 2006; Medina-Vogel et al. 2004; Ostfeld et al. 1989). Since studies of prey composition suggest that there is between-site and possibly latitudinal variation in marine otter diet (Cabello 1978, 1983; Castilla 1982; Castilla and Bahamondes 1979; Ostfeld et al. 1989), it is likely that differences in the diet, and possibly activity budgets, of otters may exist between Peru and Chile.

This otter species is forced to coexist with humans given its habitation of the marine littoral zone and exploitation of the same resources from the intertidal, subtidal and terrestrial seashore that humans exploit for food, commerce and housing (Medina-Vogel et al. 2007; Medina-Vogel et al. 2004; Moreno 2001; Moreno et al. 1984; Ostfeld et al. 1989). Thus anthropogenic impacts on marine otters are of particular concern given the continuing human population expansion and development of the Peruvian and Chilean coast which may lead to habitat loss, degradation, or fragmentation (Larivière 1998; Medina-Vogel et al. 2008).

Fishing villages are one interface between human activity and marine otter habitat (Medina-Vogel *et al*. 2007). By creating artificial den sites (*e.g.*, wharfs and shipwrecks) and providing food in the form of fish refuse, fishing villages may attract marine otters (Medina-Vogel et al. 2007). For example, groups of marine otters have been observed feeding together at fish refuse piles

(Medina-Vogel et al. 2007) which has the potential of increasing exposure to direct and indirect anthropogenic threats (Castilla 1999; Gaydos et al. 2007; Medina-Vogel et al. 2008; Medina-Vogel et al. 2004; Miller et al. 2008).

Given the paucity of information on the behavior, ecology and conservation of marine otters throughout a large part of its continental distribution, our objectives were to (1) assess marine otter diet with a view to detect any latitudinal differences, and (2) assess marine otter activity budgets and inter-specific interactions (including anthropogenic) at Peruvian fishing villages and to compare results with similar studies from Chile.

Methods

Study Area

Four distinct study sites, 54.7 to 111 km long and more than 100 km apart, along the Chilean Pacific seashore between 28.05°S and 39.67°S, and five distinct study sites in Peru, 0.2 to 21.0 km long and separated by seven to 40 km of coastline between 17.64° to 18.05°S were sampled for otter spraints (Fig. 1).

To obtain information on activity budgets and interactions with humans in Peru and for subsequent comparison with populations in Chile, otter observations were conducted from Morro Sama (17.98°S, 70.86°W) and Vila Vila (18.12°S, 70.71°W), two fishing villages in southern Peru separated by 23 km of rocky coastline (Fig. 1). Morro Sama and Vila Vila are active fishing ports with permanent human populations of approximately 200 and 350, respectively. Rats (*Rattus rattus*) as well as stray dogs and cats are common (Mangel pers obs). Both villages have developed around the placement of artificial breakwaters and fish processing facilities that have daily activity related to the arrival and departure of artisanal dive, net, and longline fishing vessels that use the ports as mooring and landing sites. Human activity is regular at both sites as are shore-based and nearshore fishing for fishes and invertebrates. These sites both possess resident groups of marine otters that use the breakwaters as den sites and forage in the vicinity of the ports.

Latitudinal variation in diet

Chilean sample sites were surveyed once between June 2005 and March 2006. For southern Peru, samples were collected at regular intervals from June 2003 to March 2009. Spraints were washed and dried at 75°C for 24 to 48 h and stored in paper bags (Bagenal 1978; Medina-Vogel et al. 2004). Spraints were analyzed for prey composition based on presence of fish or crustacean remains. Direct observations of otters feeding at the two Peruvian study sites were also made during focal follow observations (section 3.3). Prey were classified to genus or species when possible and grouped into broad categories (Table 1). Prey items that could not be identified because they were too small or were out of view, were classed as unidentified. Prey identification was guided by Chirinchigno and Cornejo (2001).

In order to compare with previous studies, data were expressed as frequency of occurrence (number of spraints in which a species occurred divided by the total number of spraints collected), and percentage of relative frequency (number of spraints in which a species occurred divided by the total occurrence of all species tested) (Medina-Vogel et al. 2004; Medina 1997; Medina 1998).

Foraging dive durations were also recorded for the Peru study sites and compared with equivalent data from Curiñanco, Chile (39°30S, 73°W) collected from December 1990 to December 1992 (Medina 1995b). Dive times were grouped and separated by site (Peru, Chile).

Human-marine otter interactions

Focal samples were collected as a method of constructing activity budgets, estimating activity bout duration and assessing human-otter interactions. Observations were conducted from the breakwaters, docks and coastline of Morro Sama and Vila Vila. Given the distance between sites and the known home range of marine otters (Medina-Vogel et al. 2007), they were considered independent. Monitoring was conducted by a total of eleven observers, with one to five observers per shift. Observers were professional biologists or trained undergraduate student volunteers. Observers used direct observation and 8x25 binoculars to aid in monitoring. Morro Sama was divided into 14 and Vila Vila into 9 continuous, non-overlapping zones approximately 50 m in length and delineated by natural or manmade markers. As the observation points selected were at locations with regular human activity related to port operation the presence of observers did not result in otter fear or avoidance behavior. Monitoring was conducted during daylight hours (6:00 to 18:00) from October 2003 through November 2007 (excluding June and August 2004, January and February 2007, July through September 2007), for a total of 701 hours at Morro Sama and 586 hours at Vila Vila.

Otter behavior was categorized as foraging, traveling, socializing (intraspecific), interacting (interspecific), resting or grooming (based upon (Shimek and Monk 1977)). Human-otter interaction in Morro Sama and Vila Vila study sites were assessed by (1) otter use intensity of the study sites zones, and (2) the level of human activity per zone. Human activity per zone was classified based upon the frequency of human use (constant daily activity, sporadic daily, infrequent) and the presence of permanent structures (docks, buildings, *etc*.) in the zone. Human activity was classified as low (no structures and infrequent activity), medium (permanent structures and sporadic daily activity) or high (permanent structures and constant daily activity).

Scan samples were collected in order to assess habitat use within the study areas and used the same activity categories and monitoring zones as focal follows. Two-minute scans were conducted every ten minutes during daylight hours (6:00 to 18:00), in March and October through December in 2003, and January through April in 2004. A total of 553 scans were conducted over nine days at Morro Sama and eight days at Vila Vila. Only scans that had full coverage by observers of all study site zones were used in this analysis.

To standardize zone sizes, aerial photographs of each site were obtained from Google Earth and zone boundaries were demarcated. Relative area utilization frequency (controlled for different zone sizes) was then determined by multiplying the total number of observations per zone by the log transformed proportional zone size. Finally, the percent of time used by otters in each zone was determined by dividing the relative area utilization frequency of each zone by the total frequency.

Statistical tests were performed using SYSTAT v.12.

Results

Latitudinal variation in diet

There was a significant latitudinal variation in the relative importance of crustaceans (GLM for unbalanced design, $F_{1,3}$=23.4; P=0.02) and fish (GLM for unbalanced design, $F_{1,3}$=30.4; P=0.01) across the latitudinal range of the study such that the proportion of fish in the diet increased from south to north while the proportion of crustaceans declined from south to north (Fig. 2). Focal observations of the marine otter diet at the Peru study sites also indicated that prey consisted mainly of fishes followed by crabs (Table 1). Of 14 identified prey species, 10 were observed more than once. The most commonly identified prey item was deep red crab *Petrolisthes desmarestii*, followed by rock crab *Cancer setosus* and rock shrimp *Rhynchocinetes typus*, but also included queen rock crab *Cancer coronatus*, sally lightfoot crab *Grapsus grapsus,* and several fish species, including: Peruvian jack mackerel *Trachurus murphyi,* Peruvian silverside *Odontestes regia,* Peruvian morwong *Cheilodectylus variagatus,* lorna drum *Sciaena deliciosa, Doydixodon laevifrons,* damselfish *Chromis spp.,* flounder *Paralichthys spp.,* and *Genypterus spp*., rays from the family Rajidae, as well as discarded Humboldt squid *Dosidicus gigas* previously used as bait.

Human-marine otter interactions

Focal follow data had equal variances (Bartlett test for equal variances P=0.52), so a general linear model (GLM) for unbalanced designs and repeated measurements was used to assess for differences in behaviors, dive time, seasons, and study sites in Peru. As there were no differences between both Peruvian study sites ($F_{7,71}$=0.83; P=0.43), data were grouped and separated by season.

Assessment of activity budgets based upon focal follows in Morro Sama and Vila Vila indicated that foraging was the most frequently observed activity at both study sites followed by traveling and that these were significantly more frequent than all other behavior categories (GLM for unbalanced design, $F_{5,138}$=198.1; P<0.01). Otters at Morro Sama spent significantly less time traveling than otters at Vila Vila (Mann-Whitney U_1=110; P=0.03), but more time foraging (Mann-Whitney U_1=22; P=0.04).

Dive durations were compared between Peruvian and Chilean study sites. Results indicate that dive times were similar between study sites (GLM for unbalanced design, $F_{3,8}$=3.6; P>0.16), with average dive times of 27.9±14.4 s (n=395) in Peru and 33.3±12.2 s (n=190) in Chile.

Marine otter habitat usage by observation zones was assessed for each study site in Peru and compared with human activity classifications. In Morro Sama, otter presence was significantly higher in areas categorized by medium human presence (Kruskal-Wallis H_2=6.2; P<0.05) but in Vila Vila, otter sightings did not vary in relation to intensity of human activity (H_2=1.781; P=0.41).

Focal follow observations of anthropogenic interactions included 113 recorded events, of which 45 (40%) were interactions with boats, and 68 (60%) were interactions with people (Table 2). This includes one instance when a marine otter was trapped in a fishing net. Of the interactions between marine otters and boats, 41 (91%) involved marine otters "searching" for food, which includes marine otters approaching boats and jumping into boats to forage for fish.

Interactions with pelicans (*Pelecanus thagus*) were frequent, with 64 events observed, including 60 instances of marine otters engaging in aggressive behavior toward pelicans. Events involving sea gulls *Larus* spp., guanay cormorants *Phalacrocorax bougainvillii*, and black-crowned night herons *Nycticorax nycticorax* were rare, with only one to two interactions observed per genus. Similarly, there was only one observed interaction between a marine otter and a South American sea lion *Otaria byronia* which did not result in any harm to the otter.

There were six observed interactions with stray cats, including two observed fights and one observed instance of an otter stealing food from a cat. One otter kill as a result of a stray dog attack was recorded in Morro Sama in September 2008. There were no observed interactions with rats, but given their nocturnal behavior this was not unexpected. Nevertheless, rats have been seen exiting the manmade breakwater at Vila Vila during storm events.

Discussion

Latitudinal variation in diet

This study has shown that fish become more prevalent and crustaceans less prevalent in the diet of the marine otter as one moves north along its distribution (Medina-Vogel et al. 2008). Crustaceans are lower quality (in terms of energy) food for otters than fish (Kruuk 1995; Medina-Vogel and González-Lagos 2008; Medina-Vogel et al. 2004). Furthermore, at the Peruvian study sites we recorded less time spent foraging than in previous studies (Medina 1995b; Ostfeld et al. 1989) and a higher percentage of fish in the diet. The higher energy content of fish compared to crustaceans (Medina-Vogel et al. 2004) may contribute to this difference in activity patterns between this and previous studies, *i.e.*, otters at our study sites may more easily fulfill their energetic needs and therefore can spend less time foraging. Thus, we postulate that capture and handling time and effort of fish prey was not sufficiently costly at our study sites to render them less "valuable" than crustaceans, as previous studies suggested (Estes 1989; Medina-Vogel et al. 2004). Hence, towards the northern part of its range, marine otter habitat seems to be of better quality in terms of prey quality which might be concentrating otters and leading to higher population densities (Medina-Vogel et al. 2007). Similar prey gradients have also been identified in other mammal species such as Eurasian otter *Lutra lutra* (Clavero et al. 2003; Remonti et al. 2009), puma *Felis concolor* (Iriarte et al. 1990) and genet *Genetta genetta* (Virgos et al. 1999).

Study results also indicated that otters at Vila Vila spent more time traveling and less time foraging than otters at Morro Sama. The greater reliance of otters at Morro Sama on lower quality shrimp prey could help account for the difference. The disparity may also be due in part to the fact that the main den locations in Vila Vila were situated further from foraging sites than in Morro Sama. Distance from dens to foraging sites has been suggested as an important habitat limitation for marine otters (Medina-Vogel et al. 2007; Medina-Vogel et al. 2008) consequently, influencing the distribution of the species as individuals balance risks and net energy gain (Buskirk 1984; Weber 1989).

Human-marine otter activity

This study found a large number of interactions between marine otters and other species, including humans. Interactions between seabirds and otters were the most frequent, perhaps due in part to competition for prey items. Otters, including marine otters, have also been shown to feed upon seabirds (de la Hey 2008; Mattern et al. 2002; Sheldon and Toll 1964; VanWagenen et al. 1981). Observed interactions between otters and anthropogenic activities support concerns over potential human impact on marine otters (Medina-Vogel et al. 2007; Medina-Vogel et al. 2008). Based upon our observations, otters appear to have adjusted to living near fishing villages in Peru, as in other fishing villages in Chile (Medina-Vogel et al. 2007). However, with otters stealing fish from nets and boats (including one occasion of an otter entangled in a fishing net) and interaction with populations of stray cats and dogs – whether through agonistic encounters, indirect competition, or disease transmission (Butler et al. 2004; Davis et al. 1972; Funk et al. 2001; Kimber and Kollias 2000) there could be detrimental effects that threaten marine otters population survival (Medina-Vogel 2010; Medina-Vogel et al. 2007). Infectious diseases, possibly associated with nearby terrestrial human development have been posited as impeding the population recovery of the southern sea otters *Enhydra lutris nereis* (Conrad et al. 2005; Johnson et al. 2009; Miller et al. 2008). Moreover, recent declines in European populations of the Eurasian otter have been attributed, in varying degrees, to anthropogenic impacts such as pollution, habitat loss, persecution and accidental mortality (Barbosa et al. 2001; Cortes et al. 1998; Macdonald 1983; Macdonald and Mason 1994; Prenda et al. 2001).

Because marine otters demonstrate apparent tolerance to living alongside fishing communities (and may actually be attracted to habitat altered by fishing communities), these anthropogenic threats could turn some of these small rocky seashore patches into population sinks. However, the overlap between marine otters and fishing villages also demonstrates the flexibility of marine otter behavior toward people. Thus, if the artificial otter habitats made by wharfs, breakwaters and shipwrecks are correctly managed in terms of otters habitat needs, they could be use a stepping stones between rocky seashore patches already separated by human dominated environments (Medina-Vogel et al. 2008; Meegan and Maehr 2002; Wikramanayake et al. 2004). The construction of a set of small artificial otter habitats along more isolated regions of rocky seashore could also be a strategy for marine otter conservation. Additional studies of marine otter ranging behavior (Medina-Vogel et al. 2007) and population densities (Medina-Vogel et al. 2006) in the northern portion of the species' range would help further characterize environmental limits for this species. The existence of a latitudinal gradient in marine otter diet also highlights the need for additional research into marine otter prey selection and the potential for overlap with commercially exploited species (Medina-Vogel et al. 2004).

Acknowledgements

We warmly thank all the volunteers that participated in this project: M. Pajuelo, J. Valqui, G. Biffi, P. Melly and D. Apaza. We also thank Dr. Rick Ostfeld for his helpful advice regarding project design in Peru. Helpful comments and suggestions on the manuscript were provided by the editor and two anonymous reviewers as well as by Dr. Lisa Ballance and Luke Gibson. Support and funding for this work was provided by Columbus Zoo and Aquarium, Darwin Initiative, Fulbright Commission, Societe D'Encouragement pour la Conservation des Animaux Sauvages, and People's Trust for Endangered Species. The marine otter project in Peru is a World Association of Zoos and Aquariums (WAZA) branded project. JCM and JAS are Overseas Research and Students Awards Scheme and University of Exeter scholarship awardees, respectively. Studies in Chile were funded by Rufford Small Grants for Nature Conservation and Project DI-01-09/R Universidad Andrés Bello. Permits for this study were provided by INRENA (No. 084-2004-IFFS-DCB).

Table 1. Marine otter prey composition at Morro Sama (MS) and Vila Vila (VV), Peru based upon focal follow observations (MS *n*=513; VV *n*=211).

	Composition of otter prey (%)	
	MS	VV
Fish	40.6	48.2
Crab	29.3	20.3
Unidentified prey	13.6	29.2
Shrimp	15.2	1.4
Squid (bait)	1.0	0
Unidentified crustacean	0.2	0
Echinoderm	0	0.5
Mollusk	0.2	0

Table 2. Frequency (%) of marine otter interactions with humans, fisheries, and other species (*n*=190) as observed during 1,392 hours of focal follows.

Interaction with:		Avoid/watch	Search/steal	Feeding	Fight/follow	Total
		Interaction type				
Human	Fishing gear	1.6	21.6	-	0.5	23.7
	Person	22.1	7.4	6.3	-	35.8
	Subtotal	*23.7*	*28.9*	*6.3*	*0.5*	*59.5*
Mammals	Stray cats	1.6	0.5	-	1.1	3.2
	Otaria byronia	0.5	-	-	-	0.5
	Subtotal	*2.1*	*0.5*	-	*1.1*	*3.7*
Seabirds	*Pelecanus thagus*	1.6	-	-	32.1	33.7
	Larus sp.	-	0.5	-	1.1	1.6
	Phalacrocorax bougainvilli	-	-	-	0.5	0.5
	Arenia sp.	-	-	-	1.1	1.1
	Subtotal	*1.6*	*0.5*	-	*34.7*	*36.8*

Figure 1. Locations of field sites for this study distributed along approximately 2,400 km of coastline in Peru and Chile. The upper inset map shows the five Peru study sites while the lower inset map shows the project area extent (black square) and the entire range of marine otters (in gray).

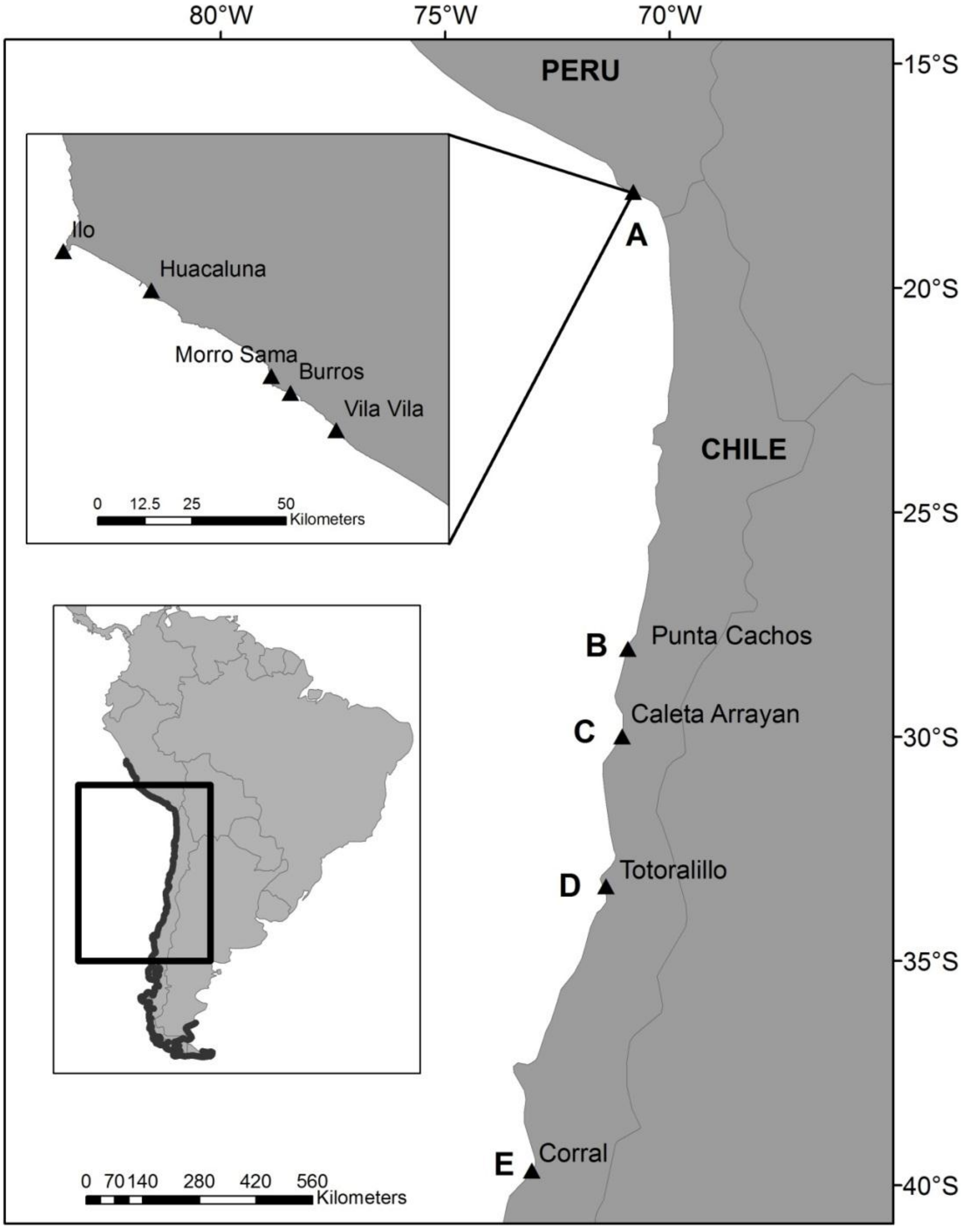


Figure 2. Diet composition given in relative frequency (RF%) by crustaceans and fish prey, found in fecal samples of marine otters collected in southern Peru (A) and four study sites in Chile (B to E). Sample size is shown for each sampled site.

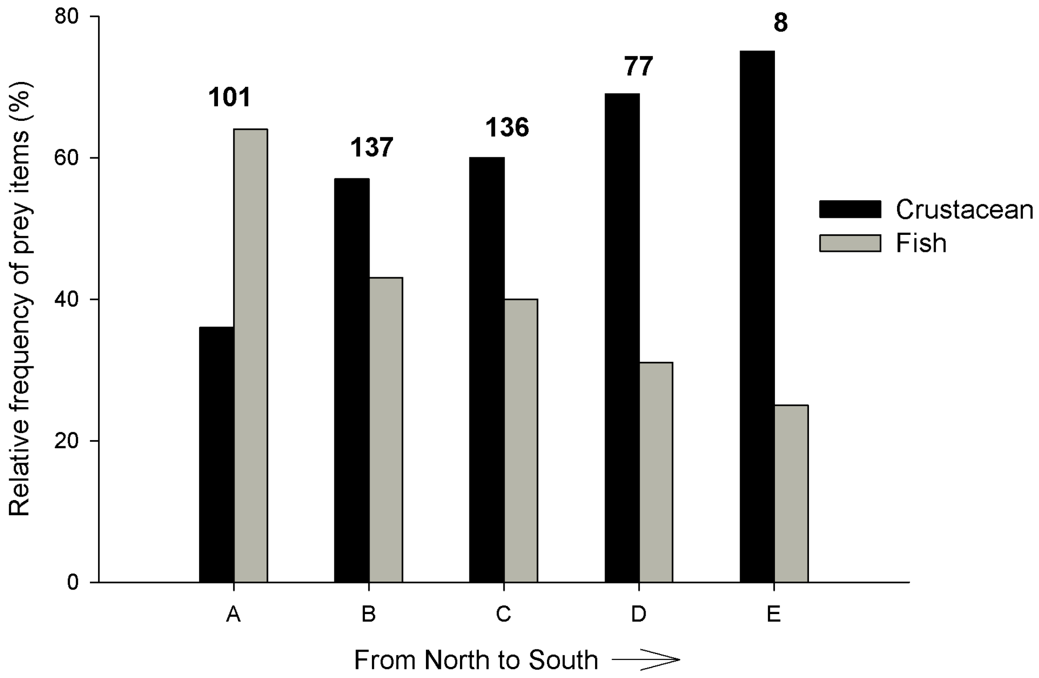
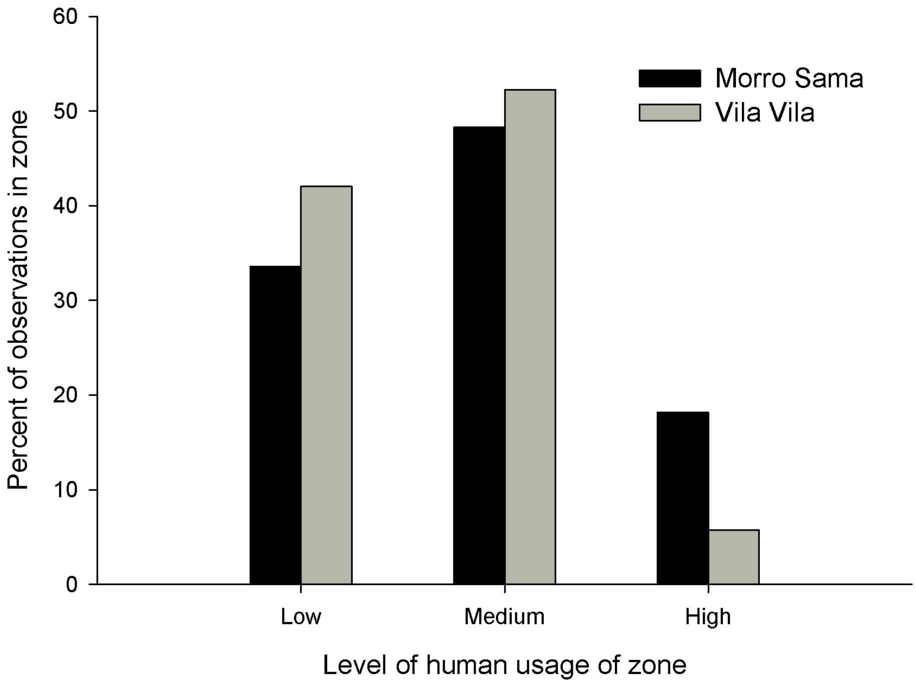


Figure 3. Habitat use by observation zones for Morro Sama and Vila Vila based upon scan sampling observations. Bars indicate the percent of total observation time otters were seen in zones grouped by level of human presence (low, medium, high).



Chapter III: Small cetacean captures in Peruvian artisanal fisheries: high despite protective legislation

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Published in Biological Conservation (2010) Volume 143: 136-143

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Abstract

We detail the first direct, at-sea monitoring of small cetacean interactions with Peruvian artisanal drift gillnet and longline fisheries. A total of 253 small cetaceans were captured during 66 monitored fishing trips (Gillnet: 46 trips; Longline: 20 trips) from the port of Salaverry, northern Peru (8°14'S, 78°59'W) from March 2005 to July 2007. The most commonly captured species were common dolphins (*Delphinus* spp.) (47%), dusky dolphins (*Lagenorhynchus obscurus*) (29%), common bottlenose dolphins (*Tursiops truncatus*) (13%) and Burmeister's porpoises (*Phocoena spinipinnis*) (6%). An estimated 95% of common dolphin bycatch was of long-beaked common dolphins (*Delphinus capensis*). Overall bycatch per unit effort for gillnet vessels (mean ± sd) was estimated to be 0.65 ± 0.41 animals.set^{-1} (range 0.05-1.50) and overall catch (bycatch and harpoon) was 4.96 ± 3.33 animals.trip^{-1} (range 0.33-13.33). Based upon total fishing effort for Salaverry we estimated the total annual average small cetacean bycatch by gillnet vessels as 2412 animals.year^{-1} (95% CI 1092 - 4303) for 2002 - 2007. This work indicates that, in at least one Peruvian port, bycatch and harpooning of small cetaceans persist at high levels and on a regular basis, particularly in driftnet vessels, despite the existence since the mid-1990s of national legislation banning the capture of marine mammals and commerce in their products. It is concluded that the coast of Peru is likely still one of the world's principal areas for concern regarding high small cetacean bycatch and there is clearly an urgent need to increase the geographic scope of observer effort to elucidate the full magnitude of this issue.

Introduction

Small scale coastal, or artisanal, fisheries make up the vast majority of global fishers, produce about half of global annual fish catch and provide most of the fish for human consumption in the developing world (Berkes et al. 2001). These fisheries are typically highly dispersed and are particularly prevalent in developing nations where regulations to monitor or manage these fisheries are frequently underdeveloped, un-enforced or non-existent (Berkes et al. 2001). Despite their size and importance, however, artisanal fisheries remain under-studied in comparison with large-scale industrial fleets (Berkes et al. 2001; Lewison et al. 2004a; Pauly 2006; Soykan et al. 2008).

Fisheries bycatch has been of growing concern in recent decades (Brothers 1991; Northridge 1991; Perrin et al. 1994). The bycatch of long-lived, late maturing, low fecundity species like marine mammals, seabirds, and sea turtles have received particular attention and it is now clear that fisheries interactions pose one of the greatest risks to the survival of many populations (Lewison et al. 2004a; Read 2008; Spotila et al. 2000). While initial attention was often primarily focused on bycatch by large industrial fleets like tuna purse-seines, and high seas driftnets (Hall et al. 2000), efforts have been intensifying to estimate the rates and evaluate the impacts of bycatch in artisanal fisheries (Alfaro-Shigueto et al. 2008; D'Agrosa et al. 2000; Moreno et al. 2006; Peckham et al. 2007; Peckham et al. 2008) and those working in coastal seas (D'Agrosa et al. 2000; Slooten 2007).

Due to their circumglobal and coastal distributions, small cetaceans are subject to human exploitation both from bycatch and direct take (Clapham and Waerebeek 2007; Jefferson and Curry 1994; Read 2008; Read et al. 2006; Reeves et al. 2003). National and international legal measures to ban the take of dolphins and porpoises in fisheries are meant to act as a protective measure to reduce declines of cetacean populations (Northridge and Hofman 1999). However, cetacean bycatch remains a concern worldwide (Lewison et al. 2004a; Read 2008; Read et al. 2006; Reeves et al. 2003). Moreover, artisanal fisheries may contribute significantly to cetacean mortality (Read et al. 2006). Gillnet fisheries in particular have been cited as probably the most significant cause globally of small cetacean mortality (Dawson and Slooten 2005; Jefferson and Curry 1994; Read et al. 2006).

Independent onboard observer programs have been widely used as an effective means to quantify bycatch (e.g. Beerkircher et al. 2002; Carretta et al. 2004; Gales et al. 1998; Rogan and Mackey 2007), and have been specifically recommended in the case of small cetacean captures in Peru (Reeves et al. 2005). Updated data on numbers of cetaceans caught and the spatio-temporal distribution of cetaceans and bycatch are essential in defining the scale of any problem and in designing appropriate national and regional management strategies (Reeves et al. 2005; Reeves et al. 2003). Moreover, the IUCN Cetacean Specialist Group (CSG) and the IWC Scientific Committee have both listed the Peruvian dusky dolphin and Burmeister´s porpoise as priorities for cetacean bycatch reduction.

In Peru, previous research into small cetacean captures has focused on the monitoring of landings of carcasses and fishmarkets for the presence of small cetacean products (Garcia-Godos 1992, Majluf et al. 2002; Read et al. 1988; Van Waerebeek 1994; Van Waerebeek et al. 2002b; Van Waerebeek et al. 1997b; Van Waerebeek and Reyes 1994a; Van Waerebeek and Reyes 1990). Captures of small cetaceans were thought to have peaked in the period 1990-1993 when estimates of total take by artisanal and commercial fisheries ranged between 15 000 and 20 000 animals per annum (Van Waerebeek and Reyes 1994a), making it one of the largest small cetacean takes in the world. Ministerial decrees (1990 and 1994) reinforced by a national law in 1996 (Anonymous 1996), prohibit the intentional take, landing and sale of small cetaceans in Peru (reviewed in Van Waerebeek et al. 1994), but this legislation is not fully enforced and the capture and trade of small cetaceans continues (e.g. Van Waerebeek et al. 2002b). The legislation did, however, have the effect of reducing reported landings and pushing the continuing trade in small cetaceans into the black market which was much more difficult to monitor (Van Waerebeek et al. 2002b; Van Waerebeek et al. 1997b). In addition it was expected that, unlike before, at least some fishermen would simply discard cetacean bycatch offshore so as to avoid any problems with landings of legal fish catches. As a result, other methods are required to quantify the continuing catch of small cetaceans. Here we report on recent at-sea observations of artisanal gillnet and longline activities allowing the first direct effort-corrected estimates of bycatch for artisanal fisheries operating from an important Peruvian port.

Material and methods

Onboard observer scheme

From March 2005 to July 2007 observers monitored a total of 66 artisanal fishing trips (480 sets; 439 fishing days) for small cetacean bycatch. Artisanal fisheries are defined here, according to Peruvian fisheries regulations, as containing boats with a maximum of $32.6m^3$ of storage capacity, less than 15m of length, and principally based on the use of manual work during fishing operations (Ley General de Pesca 2001). Trips monitored were on gillnet and longline vessels originating from the port of Salaverry (8^{o}14'S, 78^{o}59'W), an artisanal port in northern Peru and home to over 100 fishing vessels (Alfaro-Shigueto et al., unpublished results). Skippers (N=21) upon whose vessels observers operated were voluntary participants in the project. Observers did not take part in fishing activity. Observers worked in all months of the year over a total period of 29 months, in order to account for possible seasonal variation in magnitude and spatial patterns of effort.

At-sea observers

All observers were biologists and were trained in relevant data collection methods including marine mammal identification. Data were gathered on specific gear used (longline or gillnet), the timing and position (using GPS) of each set and any bycatch occurring. All observers were equipped with cameras and photographed unusual or unidentifiable captures for later species identification. Common dolphins *Delphinus* spp. were not identified to species in the boats, nor were *T. truncatus* assigned to inshore/offshore morphotype, considering there was a degree of uncertainty about positive identification among observers.

Photos of *Delphinus* spp. (n=38) examined by the authors indicated bycatch of thirty-six long-beaked common dolphins *D. capensis* (94.7%) and two short-beaked common dolphins *Delphinus delphis* (5.2%). This composition estimate is used in our extrapolation to the wider estimate of take (Table 4). The overwhelming preponderance of *D. capensis* found here is broadly consistent with the more than 99% of *Delphinus* catches belonging to *D. capensis* in Peru based on a sample of 1,067 common dolphins taken in coastal fisheries in the period 1984-1993 (Van Waerebeek 1994).

Shore-based observers

Shore-based observers were employed in Salaverry to monitor daily fishing activity from September 2001 to March 2008. Observers collected data on the total number of fishing trips departing and returning per day and per vessel type, locations of fishing activity and associated catch and bycatch. Data collection was based upon daily interviews with fishermen and monitoring of dockside activity. Respondents were informed that the information would be kept anonymous and be used strictly for research purposes. Fishermen returning from fishing trips were queried regarding vessel type, fishing effort, target catch, and incidents of bycatch of small cetaceans, sea turtles or seabirds. Resulting data therefore are a census of fishing effort by gear type over the study period.

Data analysis

All observer data were managed in a Microsoft Access database. Bycatch per unit effort (CPUE) was calculated on per trip and per set basis for both fishing gears. For gillnet vessels, CPUE was also presented per length (km) and area (km^2) of net set. Descriptive statistics are presented as mean ± standard deviation (SD) or with 95% confidence intervals (CI) unless specified otherwise. Statistical tests were performed using SPSS 15.0 and Genstat 10. For temporal analyses of total bycatch we used General Linear Models (GLMs) with normal errors, where CPUE was the dependent variable with season and year as factors. In this instance CPUE was calculated on a trip by trip basis by dividing the number of bycatch incidents by the number of sets made. When it came to analysis of the bycatch for individual species the date distributions departed significantly from normality and there were significant differences in variances among groups. We therefore used raw count data as our dependent variable, with number of sets included as a covariate (to account for variation in effort across seasons/years) along with season and year as factors. We also employed GLMs for these analyses but fitted them with Poisson errors and a log link function. Season was divided as follows: season 1 = Nov-Jan; season 2 = Feb-Apr; season 3 = May-Jul; season 4 = Aug-Oct. All spatial analyses and maps were prepared using ESRI ArcMap 9.1, MATLAB 7.6 and Hawth's Tools (Beyer 2004). Bathymetry values were determined with Global Gridded Relief Data (ETOPO2v2) with 2' minute resolution (USDOC 2006). Quartic kernal and 50% and 75% probability contour analyses were performed using 2km grid spacing and least squares cross validation derived optimized smoothing factors for longline and gillnet sets (25km) and a smoothing factor of 35km for small cetacean capture locations.

Estimating bycatch rates and total bycatch

Gillnet bycatch data for the study were grouped by month in order to derive monthly stratified CPUE estimates. These rates were calculated in terms of catch.trip^{-1}, catch.set^{-1}, and to facilitate comparison with other studies, catch.km of net length^{1}, and catch.km^{2} of net area^{-1} were also calculated. Given the small sample size we did not prepare similar monthly stratified catch estimates of longline bycatch.

Based upon the catch rates derived in this study and the data on monthly Salaverry fishing effort from 2002 to 2007, we were able to estimate the number of small cetacean captures (overall and per species) for the gillnet fleet. To derive these values we applied the monthly CPUE rates calculated in this study to the estimated number of monthly gillnet sets for the years 2002 to 2007. Monthly number of sets was estimated by multiplying the known number of trips per month by the average number of sets per trip as determined by this study. Month specific CPUE calculations were used to generate an estimate. Bycatch data for each individual month were pooled to derive monthly CPUE values. As bycatch data were left skewed, the monthly bycatch estimates were calculated by multiplying the CPUE of sets with bycatch by the total number of sets multiplied by the proportion of sets in that month estimated to have bycatch (as determined in this study). Monthly catch estimates for each year were then summed to arrive at annual totals.

Results

Gillnet characteristics

This project monitored 46 trips (341 sets; 319 fishing days) by artisanal drift gillnet vessels (Table S1). A detailed summary of trip and net characteristics is presented in Table 1. All monitored trips targeted sharks and rays (mainly smooth hammerheads (*Sphyrna zygaena*), eagle rays (*Myliobatis spp.*), blue sharks (*Prionace glauca*), short-fin mako sharks (*Isurus oxyrinchus*) and thresher sharks (*Alopias vulpinus*)). Gillnets observed were made of multifilament nylon cord of varying mesh sizes. Nets were set at the ocean surface and were typically set in the afternoon and retrieved the following morning. The average number of sets per trip was 7.4 ± 2.4 (range: 2-11). Total net length per set averaged 1948 ± 512m (range: 1097-3072). The only observed bait used was small cetacean blubber or meat.

Longline characteristics

A total of 20 trips by artisanal longline vessels (138 sets; 167,670 hooks; 129 fishing days) were monitored (Table S1). Sixteen of 20 trips (80%) targeted dorado (*Coryphaena hippurus*) with the remaining 4 trips targeting sharks (mainly blue and short-fin mako). Mainlines for all trips were set at the sea surface and were made of multifilament nylon rope. While trip lengths were similar, vessels targeting sharks typically had more and longer sets and deployed fewer, more widely spaced hooks than vessels targeting dorado. Branchlines were made of narrow diameter nylon multifilament cord, with branchline length of vessels fishing for sharks approximately double that of vessels fishing for dorado. Leader material used was either nylon monofilament when targeting dorado or metal cable when targeting sharks. Jumbo flying squid (*Dosidicus gigas*) was used as bait for both sharks and dorado while small cetacean blubber and meat was also used as bait by vessels targeting sharks.

Summary of small cetacean interactions

A total of 253 dolphins and porpoises were observed by onboard observers as captured during the study period (Table 1). Bycatch in gillnets accounted for 91.3% of all interactions recorded with another 6.3% (n=16), 2.0% (n=5) and 0.4% (n=1) coming from longline harpooning, gillnet harpooning and longline bycatch, respectively.

Gillnets

Eighty percent of gillnet trips (37 of 46 trips; 104 sets) experienced small cetacean bycatch and the majority of captures were of two species (common dolphins 50.2%; dusky dolphins 27.7%). Captures also included common bottlenose dolphins (13.0%; n=30), Burmeister's porpoises (6.9%; n=16), Risso's dolphins (*Grampus griseus*) *(*0.4%; n=1*)*, and unidentified small cetaceans (1.7%; n=4; Table 2). Mean CPUE of small cetaceans was 0.65 ± 0.41 animals.set^{-1} (Range: 0.05-1.50) or 4.96 ± 3.33 animals.trip^{-1} (Range: 0.33-13.33) (Table S2). In addition to bycatch, 3 common bottlenose dolphins and 2 common dolphins were harpooned for bait on three gillnet fishing trips by three different vessels.

Longlines

Small cetacean bycatch was only observed on one (5%) longline fishing trip by a vessel targeting sharks and using small cetacean meat as bait. The bycatch was of a dusky dolphin, the branchline having been entangled around its flukes/tail stock. In addition, however, on 3 of 4 longline trips targeting sharks (15% of total observed trips), dolphins were harpooned for bait (Table 1). While we did not prepare monthly stratified catch estimates, the overall interaction rate for longline vessels targeting sharks was a relatively high 4.25 ± 3.86 animals.trip^{-1} (Range: 0-8) due to the common practice of harpooning dolphins for bait.

Fates of captured cetaceans

All harpooned animals, both by gillnet and longline vessels, were used as bait (Table 1). Also, the one dusky dolphin bycaught by a longline vessel, while captured alive, was killed and used as bait. Twenty-nine percent of gillnet bycatch was used as bait, including 54.7% of dusky dolphins. Ninety-seven percent of gillnet entangled animals were recovered dead. Of these, the most frequent fate of the carcass was for it to be discarded at sea (39.8%). Half of all common dolphins (50.0%) and a similar proportion of common bottlenose dolphins (56.7%) bycaught in gillnets were discarded dead. Gillnet entangled animals were also used for bait on subsequent sets during the trip, later sold in local markets, consumed on the boat or at home, released alive, or were given or sold to other gillnet or longline vessels for use as bait. Although constituting a small part of the total, there is a suggestion that Burmeister's porpoises may be preferred for human consumption, with 71.5% of known fate animals either consumed by the boat crew or brought to shore to be eaten at home.

Spatial distribution

The scarcity of reliable bathymetry data in coastal zones (<200m) makes detailed interpretation of the depths of captures difficult since most captures were in less than 250m depth (Cracknell 1999; Malthus and Mumby 2003). However, several general patterns do emerge when examining fishing effort and small cetacean capture locations. Gillnet sets were more coastal than longline sets (Fig. 1a,b) with gillnet trips occurring over the continental shelf and longline trips occurring on the continental slope or pelagic. All small cetacean interactions appear to take place on the continental shelf or near the slope (Fig. 1c). All harpooning and longline bycatch events occurred within their respective 90% probability contours of set locations. There was a statistically significant difference in perpendicular distance to shore of captures among the four most commonly taken species (H=42.9, df=3, Kruskal-Wallis, P<0.001), with captures of Burmeister's porpoises significantly nearer to shore than other species, occurring in a small area fronting Salaverry (Fig. S1a-d).

Temporal Distribution

Total bycatch per trip varied seasonally ($F_{3,42}$ = 4.4, p=0.009) but not annually, nor was there a significant interaction between season and year (Fig. S2a). Post hoc Scheffe tests indicate that total bycatch in season 4 (Aug-Oct) was much higher than in other months (means ± SE: season 1 = 0.19 ± 1.9; season 2 = 0.61 ± 0.14; season 3 = 0.77 ± 0.15; season 4 = 1.18±0.2). In the case of common dolphin bycatch, both season (Wald =11.12, df= 3, p =0.11) and year (Wald = 8.75, df=2, p=0.13) have significant effects, with number of sets and the season x year interaction having no significant influence (Fig, S2b). In this instance, pairwise comparisons indicate that season 1 has lower bycatch than other seasons (p<0.05.Estimated marginal means (EMMs) ± SE: season 1 = 0.67 ± 0.28; season 2 = 1.94 ± 0.38; season 3 = 2.59 ± 0.48; season 4 = 1.66 ± 0.44) and that bycatch in 2007 was much lower than in 2006 and 2005 (p<0.05. EMMs ± SE: 2005 = 2.23 ± 0.43; 2006 = 2.39 ± 0.35; 2007 = 0.67 ± 0.28). Only year had a significant effect on dusky dolphin bycatch (Wald =46.1, df = 3, p<0.001). In this instance the highly significant effect is driven by multiple differences among seasons. Dusky dolphin bycatch was significantly lower in seasons 1 (EMM ± SE = 0.13 ± 0.12) and 2 (0.44 ± 0.17) than in seasons 3 (1.5 ± 0.33) and 4 (4.37 ± 0.74) (Fig. S2c). In contrast, the bycatch of bottlenose dolphins (Fig S2d) show no strong seasonal patterns but year (Wald =7.5, df=2, p=0.025) and total sets (Wald=20.2, df=1, p<0.001) are both significant, with higher bycatch in 2007 (EMM ± SE = 0.97 ± 0.37) than in 2006 (0.32 ± 0.11) and 2005 (0.34 ± 0.14). For Burmeister's porpoises (Fig. S2e), there were no bycatch incidents observed in 2006 and 2007 so year is not included in the analysis. The bycatch of this species shows a very weak seasonal effect (Wald=6.2, df=2, p=0.047) with lower bycatch in seasons 2 (EMM ± SE = 0.40 ± 0.28) and 3 (0.33 ± 0.33) than in seasons 1 (2.00 ± 0.70) and 4 (2.00 ± 0.70). However, the sample size for this last species is small and thus these results should be treated with some caution.

Estimating Annual Totals

Based upon daily shore-based monitoring of fishing effort in Salaverry we determined that there were an average of 518.2 ± 90 gillnet trips (range: 411 to 620 trips.year^{-1}) and 300.7 ± 25.2 longline trips (range: 272 to 341 trips.year^{-1}) per annum, for the years 2002 to 2007 (Table 2; Fig. 2). For the years 2002 to 2007 the estimated annual number of small cetaceans bycaught by gillnet vessels in the port of Salaverry was 2412 (95% CI 705-4415). The number of small cetaceans harpooned by gillnet vessels was estimated to be on the order of some tens of animals.

Discussion

The work presented here provides the first direct, at-sea monitoring of small cetacean interactions with Peruvian artisanal gillnet and longline vessels. It has shown that, in at least one port in northern Peru, a sizeable level of bycatch, direct take through harpooning, and consumption of small cetaceans, continue despite the existence since the mid-1990s of national legislation banning the capture of marine mammals and commerce in their products. Previous work monitoring the take of small cetaceans in Peru's artisanal fisheries focused largely on dockside monitoring of landing, monitoring of fishmarkets for small cetacean products and assessing beach cast carcasses for evidence of fishery interactions (Read et al. 1988; Van Waerebeek 1994; Van Waerebeek et al. 2002b; Van Waerebeek et al. 1997b; Van Waerebeek and Reyes 1990). Take observed here consisted of the same species assemblage documented in previous market studies (Read et al. 1988; Van Waerebeek et al. 2002b; Van Waerebeek et al. 1997b; Van Waerebeek and Reyes 1994a; Van Waerebeek and Reyes 1990).

The magnitude of the issue

Our results indicate that, for this site, bycatch in gillnets is the main cause of mortality with CPUE higher than published accounts from the California driftnet fleet off the United States Pacific coast (Barlow and Cameron, 2003), the Spanish driftnet fleet in the western Mediterranean (Silvani et al. 1999), and Ecuadoran artisanal gillnets (Felix and Samaniego 1994) and comparable to CPUE for the large scale Moroccan driftnet fleet in the southwest Mediterranean (Tudela et al. 2005). CPUE in the Irish driftnet fleet in the northeast Atlantic was higher than observed here (Rogan and Mackey 2007), however total annual estimated catch was about half that we have estimated for the port of Salaverry. As in other studies both in Peru (Ilo; Alfaro-Shigueto unpublished results), and in the southern ocean (Kock et al. 2006), South Georgia (Ashford et al. 1996), and Hawaii (Forney and Kobayashi 2007) cetacean bycatch rates at the vessel level in longlines were considerably lower than those of gillnet vessels. The overall interaction rate for longline vessels in Peru has the potential to be high, however, given the frequency of harpooning observed in Salaverry (3 of 4 trips targeting sharks).

The Peruvian artisanal fleet has more than doubled in size from 1997-2005 to 9667 vessels and vessels in the port of Salaverry represent only ca. 1% of that fleet and ca. 2% of gillnetters (Escudero 1997; Estrella 2007; Estrella et al. 1999; Estrella et al. 2000). It is feasible therefore, that at the national level, interactions between artisanal fisheries and small cetaceans remain globally significant. Indeed, it is conceivable that total mortality by the artisanal fishery is of the order or greater than that estimated in 1990-1993 (15,000 to 20,000 small cetaceans annually for all of Peru, Van Waerebeek and Reyes 1994a). An annual catch rate of this magnitude would be one of the highest estimated takes globally, on the order of that reported for Japan, Sri Lanka, or the large scale Morrocan driftnet fleet (Bjorge et al. 1991; Leatherwood 1994; Reeves et al. 2003; Tudela et al. 2005). For the port of Salaverry alone, our estimate of small cetacean captures is approximately equivalent to that of all recorded fisheries in the United States of America (Read et al. 2006).

Challenges to and opportunities for take reductions

Almost all gillnet bycatch was recovered dead and approximately 40% of all entangled small cetaceans were discarded at sea. Thus, while 60% of carcasses were used opportunistically as bait or for consumption, the fact that the other 40% of all bycatch was discarded indicates that interactions with small cetaceans are often unwanted. This also points to the mixed success of Peru's protective legislation. That legislation succeeded in reducing landings and shrinking the market for small cetacean products, but does not appear to have reduced small cetacean captures at sea. Current practice stands in sharp contrast with the 1985-1994 situation when discards were rare and most carcasses were landed to be sold, openly or covertly (e.g. Van Waerebeek and Reyes 1994a). This suggests that the promotion and implementation of bycatch avoidance measures in the gillnet fishery may now, perhaps for the first time, be acceptable to fishermen as a means of reducing unwanted catch. Given prevailing levels of poverty, the extent and size of the fishery and resource available for natural resource management, closure of fishing areas to gillnetting or modification of gillnets (Dawson 1991) appear unimplementable. The use of acoustic alarms has been shown to have potential in reducing gillnet bycatch in some cetacean populations (Barlow and Cameron 2003; Cox et al. 2003; Kastelein et al. 2001; Koschinski et al. 2006; Kraus et al. 1997; Leeney et al. 2007) and should be trialed in the Peruvian gillnet fishery.

Clearly though, a demand for small cetacean products in the form of bait and meat persists. Bait was collected from entangled animals but also from animals harpooned specifically to collect bait. Harpooning for bait occurred on both gillnet and longline vessels. When used in gillnets, pieces of dolphin blubber and meat were tied to the center of each net pane. Dolphin blubber and meat was the only bait observed used in gillnets during the study and was used specifically due to its claimed effectiveness in attracting blue and short-fin mako sharks. Use of small cetaceans as bait was also reported during interviews with fishermen (both in Salaverry and Pucusana) where they noted dolphin meat's particular effectiveness for catching sharks given its high blood and fat content and its characteristic, unlike some fish bait, to remain intact and attached even after extended periods of soaking (this study; Van Waerebeek, unpublished results). Previous work also reported this usage and warned that increasing demand for small cetacean meat and blubber as shark bait could offset any reductions in small cetacean take as a result of the ban on capture and commerce (Van Waerebeek et al. 2002b; Van Waerebeek et al. 1997b). The use of small cetaceans as bait has also been reported in coastal communities inter alia in Colombia (Avila et al. 2008; Mora-Pinto et al. 1995), Argentina (Goodall et al. 1994), Chile (Lescrauwaet and Gibbons 1994), Mexico (Zavala-Gonzalez et al. 1994) and the Philippines (Dolar 1994), but the practice is common worldwide. In discussions with fishermen during this study regarding their use of small cetaceans for bait, a large number indicated that one reason for the use of dolphins and porpoises was the high cost of their preferred traditional bait fishes like mackerel (*Scomber japonicus*). Although challenging, finding an appropriate, low-cost substitute bait to cetacean meat and blubber may reduce harpooning. This is particularly urgent given recent evidence that the practice of harpooning small cetaceans for use as longline bait is prevalent along the entire Peruvian coast, most recently being reported in the southern port of Ilo in November 2008 (Bernedo, personal communication).

Future directions

The current study makes clear that small cetacean bycatch and direct take continues despite the existence of national legislation prohibiting capture and commerce in their products. Our results mandate renewed interest on the part of all stakeholders to expand the scope of research and monitoring of small cetacean populations and their interactions with Peru's artisanal fleet. Our study demonstrates the feasibility and use of independent observer programs onboard artisanal fishing boats, and we strongly recommend that such surveys be continued and expanded throughout all fisheries of concern across the full geographic scale. Priority should be given to increased monitoring of gillnet fisheries in the center and north of the country where the fleet is concentrated. Given the large number of ports and landing sites used by the artisanal fleet it may be more practicable to choose a number of 'index' ports distributed along the coast and to focus on maximizing onboard observer coverage in these locations. Observer effort should optimally be continuous in order to account for any temporal variations in interactions, or should at least ensure an adequate coverage of all seasons. Special attention should be paid to interactions with dusky dolphins and Burmeister's porpoises since previous research indicate that the Peruvian populations of these species form reproductively and genetically isolated stocks that should be subject to stock specific management measures (Cassens et al. 2003; Cassens et al. 2005; Rosa et al. 2005; Van Waerebeek 1992, 1993).

While large, the artisanal fishery is one of several fisheries operating in Peruvian waters and potentially interacting with small cetaceans. One must also consider interactions with other fisheries, most notably industrial and artisanal purse-seine vessels targeting small schooling fish, especially anchovy. Based upon onboard observer effort of 2% of the industrial fleet in 2002, van Oordt and Alza (2006) reported an average capture rate of 0.041 $dolphins.set^{-1}$. They noted that small cetacean captures in the fishery could be significant given the estimated 80 000 fishing trips per year. Data on fishing effort for all fisheries operating in Peru's coastal waters need to be compiled in order to more effectively set the research agenda towards building a clearer understanding of the possible impacts on small cetacean populations.

Acknowledgements

We kindly acknowledge the fishermen and field observers for their generous collaboration. Pro Delphinus staff biologists and veterinarians P. Melly, M. Pajuelo, and D. Vega provided superb data and field work. We also extend our thanks to the editor and two anonymous reviewers who provided important comments and suggestions on the manuscript. Pro Delphinus (JCM, JAS, CCB) research on cetacean bycatch has received funding from the Oak Foundation though Duke University, Rufford Foundation, Whale and Dolphin Conservation Society, Cetacean Society International and Cleveland Metroparks Zoo. CEPEC (KVW) research on cetacean bycatch has benefitted from long-term International Fund for Animal Welfare (IFAW) support. JCM and JAS are Overseas Research and Students Awards Scheme (ORSAS) and University of Exeter scholarship awardees, respectively. SB, BJG and MJW receive funding from Darwin Initiative, Natural Environment Research Council and the South West Regional Development Agency through the Peninsular Research Institute for Marine Renewable Energy.

Table 1. Species composition, capture methods and use of small cetacean carcasses of all interactions (n(%)) with gillnet and longline vessels. Percentages are read across for capture methods and uses while species composition of gillnet bycatch subtotal and grand total (2nd and last columns from left) are tallied by column. The fate category "unknown" refers to animals for which final fate was not recorded.

Species	Grand Total	Longline harpoon	Longline Bycatch	Gillnet harpoon	Gillnet bycatch: For bait	Discarded dead	Released alive	Sold	Eaten (boat)	Eaten (home)	Unknown	Gillnet subtotal
Delphinus spp.	120 (47)	2 (1.7)	0	2 (1.7)	22 (18.3)[a]	58 (48.3)	1 (0.8)	10 (8.3)	0	5 (4.2)	20 (16.7)	116 (50.2)
L. obscurus	73 (29)	8 (11.0)	1 (1.4)	0	35 (47.9)[b]	15 (20.5)	1 (1.4)	4 (5.5)	1 (1.4)	1 (1.4)	7 (9.6)	64 (27.7)
T. truncatus	33 (13)	0	0	3 (9.1)	10 (30.3)	17 (51.5)	1 (3.0)	0	0	0	2 (6.1)	30 (13.0)
P. spinipinnis	16(6)	0	0	0	1 (6.3)	1 (6.3)	0	0	2 (12.5)	3 (18.8)	9 (56.3)	16 (6.9)
Unidentified	10 (4)	6 (60.0)	0	0	0	0	0	0	0	0	4(40.0)	4 (1.7)
G. griseus	1 (0.4)	0	0	0	0	1(100)	0	0	0	0	0	1 (0.4)
Total	253 (100)	16 (6.3)	1 (0.4)	5 (2.0)	68 (26.9)	92 (36.4)	3 (1.2)	14 (5.5)	3 (1.2)	9 (3.6)	42 (16.6)	231 (100)

[a] 4 animals sold to longline vessel while at sea.

[b] 2 animals given to another gillnet vessel and 2 stored for use on a subsequent longline trip.

Table 2. Estimated annual bycatch of small cetaceans by gillnet vessels for the port of Salaverry for the years 2002 - 2007, mean (95% CI). Values are derived from annually pooled monthly estimates of bycatch and known levels of monthly fishing effort for the port. Presented estimates are of total estimated small cetacean captures and of the four most commonly captured species.

Year	# trips	Estimated # sets	Total estimated bycatch	*D. capensis*	*L. obscurus*	*T. truncatus*	*P. spinipinnis*
2002	411	3054	2002 (845-3776)	690 (431-1011)	812 (189-1979)	189 (112-266)	191 (94-769)
2003	620	4607	3212 (1356-6047)	1168 (709-1713)	1284 (279-3188)	311 (187-435)	263 (148-982)
2004	421	3128	2118 (945-3839)	825 (437-1334)	759 (155-1845)	213 (98-328)	183 (86-629)
2005	572	4250	2518 (1247-4323)	1186 (680-1892)	619 (173-1368)	303 (129-477)	237 (134-677)
2006	593	4406	2636 (1278-4505)	1158 (619-1931)	773 (216-1719)	285 (129-441)	228 (115-756)
2007	492	3656	1987 (881-3330)	814 (372-1418)	662 (156-1666)	255 (158-352)	129 (56-385)
Average	518	3850	2412 (1092-4303)	973 (541-1550)	818 (195-1961)	259 (136-383)	205 (105-699)

Table S1. Summary of monitored fishing gears and their characteristics. Mean±SD (Range).

Category	Gear types and characteristics		
Fishing gear	Longline		Driftnets
Target species	Sharks	Dorado	Sharks and rays
Trips	4	16	46
Sets	29	109	341
Trip duration (days)	9.0±1.2 (8-10)	9.6±3.1 (5-14)	8.2±2.4 (3-13)
Fishing days	29	106	319
Set depth	Surface	Surface	Surface
# sets.trip^{-1}	7.0±0.8 (6-8)	6.3±2.4 (2-11)	7.4±2.4 (2-11)
Set length (hours)	11.0±3.5 (2.1-18.7)	8.7±2.1 (2.4-16.1)	14.5±5.0 (0.78-38.6)
# hooks	911±295 (600-1200)	1291±199 (1000-1650)	na
Branchline length (m)	11.4±6.0 (5.5-18.3)	5.75±0.75 (4.6-7.3)	na
Total hooks observed	25,500	142,170	na
# panels	na	na	20.7±4.4 (10-36)
Panel length (m)	na	na	95.9±23.7 (55.9-146.3)
Panel height (m)	na	na	12.1±1.5 (9.1-14.6)
Net length per set (m)	na	na	1948±512 (1097-3072)
Net area per set (km^2)	na	na	0.024±0.007 (0.01-0.04)
Net deployed per trip (km)	na	na	14.5±6.6 (3.7-30.1)
Net deployed per trip (km^2)	na	na	0.17±0.08 (0.04-0.38)
Total net observed (km)	na	na	665.6
Total net observed (km^2)	na	na	8.01
Mesh size (cm)	na	na	11.2 to 25.4
Hook type	J1, J2, J4, J5	J4, J5	na
Bait type	Small cetacean / squid	Squid	Small cetacean
Set time	Morning	Morning	Afternoon
Haul time	Afternoon (same day)	Afternoon (same day)	Morning (next day)

Table S2. Gillnet total catch (bycatch and harpoon combined) and bycatch CPUE of all small cetaceans combined and of the four most commonly captured species. To facilitate comparison with other studies results are presented as catch. trip^{-1}, catch .set^{-1}, catch.km of net^{-1}, and catch .km^{2} of net area^{-1} but see Methods section 2.5 for a detailed description of the procedure for estimating total bycatch presented in Table 2. Values are averages of pooled monthly catch and bycatch CPUE observed during the study period, March 2005-July 2007. Total catch estimates are not presented per set, per km and per km^{2}, because harpooning occurs independent of fishing effort at the set level.

	Per trip		Per set		Per km		Per km^{2}	
Species	Mean±SD	Range	Mean±SD	Range	Mean±SD	Range	Mean±SD	Range
Total catch	4.96±3.33	0.33-13.33	na	na	na	na	na	na
Overall bycatch	4.84±3.23	0.33-13.00	0.65±0.41	0.05-1.50	0.40±0.32	0.03-1.15	32.29±25.70	2.38-91.53
Delphinus spp.	1.99±1.60	0-5.29	0.26±0.20	0-0.63	0.13±0.10	0-0.26	10.60±8.15	0-24.77
L. obscurus	1.77±2.46	0-8.67	0.25±0.35	0-1.00	0.17±0.25	0-0.72	14.46±21.28	0-57.74
T. truncatus	0.52±0.52	0-1.33	0.06±0.82	0-0.22	0.04±0.05	0-0.14	2.95±3.89	0-10.79
P. spinipinnis	0.38±0.50	0-1.67	0.06±0.09	0-0.25	0.04±0.06	0-0.20	2.84±4.39	0-13.48

Figure 1. Set locations by (a) gillnet vessels, (b) longline vessels, and (c) of all gillnet bycatch. Also presented in each pane are 50% and 75% probability contours of fishing sets and gillnet bycatch (250m, 750m, 2,000m and 3,000m isobaths are indicated).

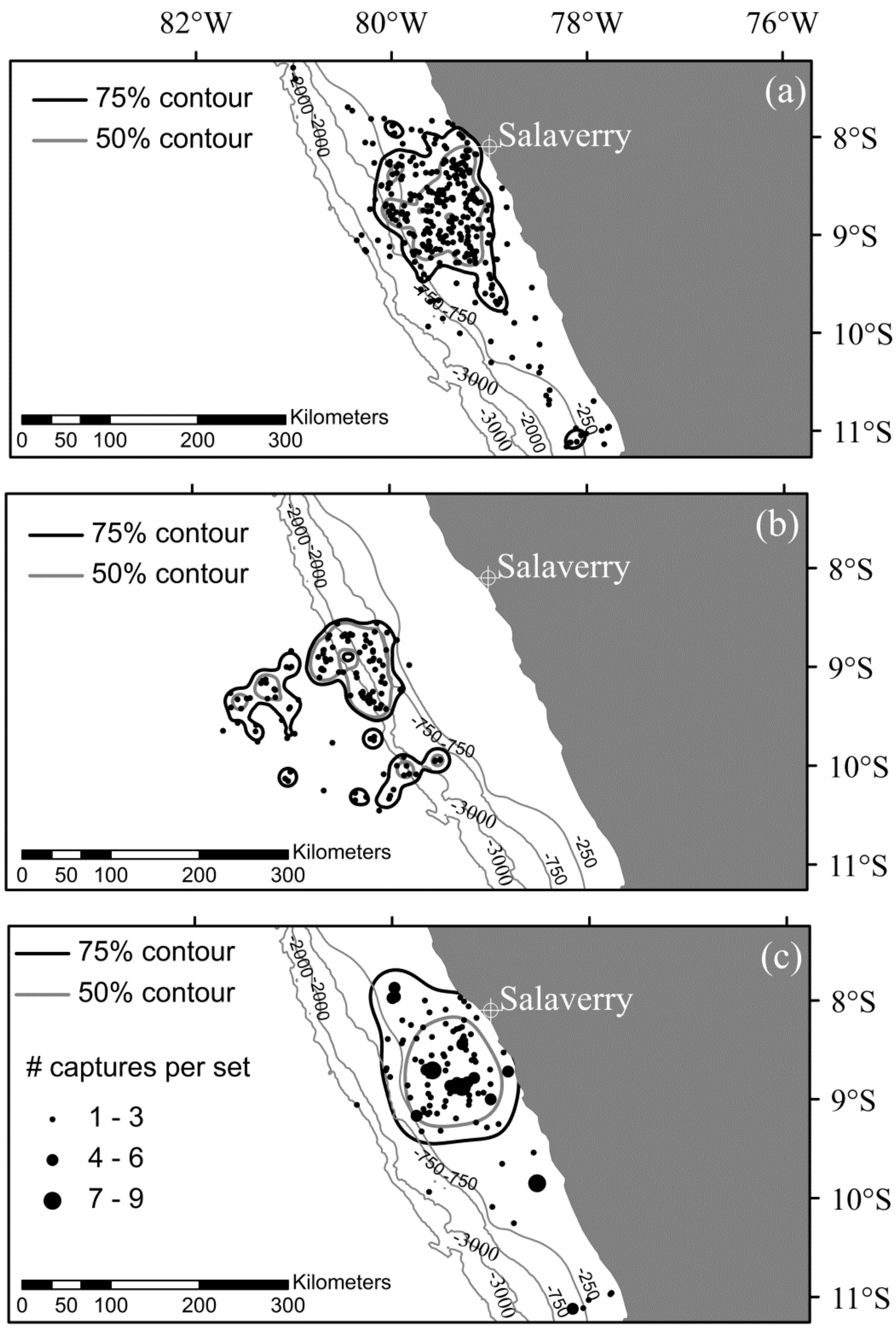


Figure 2. Monthly average number of trips by gillnet and longline vessels for the years 2002 - 2007 determined from.daily dock-side monitoring of fishing activity.

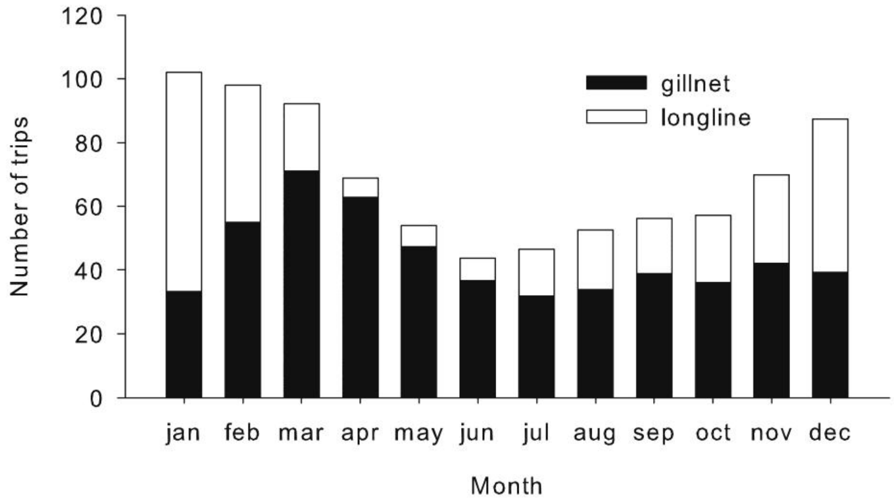


Figure S1. Set locations of gillnet bycatch of (a) *Delphinus* spp., (b) *L. obscurus*, (c) *T. truncatus,* and (d) *P. spinipinnis*. Also presented in each pane are 50% and 75% probability contours of fishing sets and gillnet bycatch (250m, 750m, 2,000m and 3,000m isobaths are indicated).

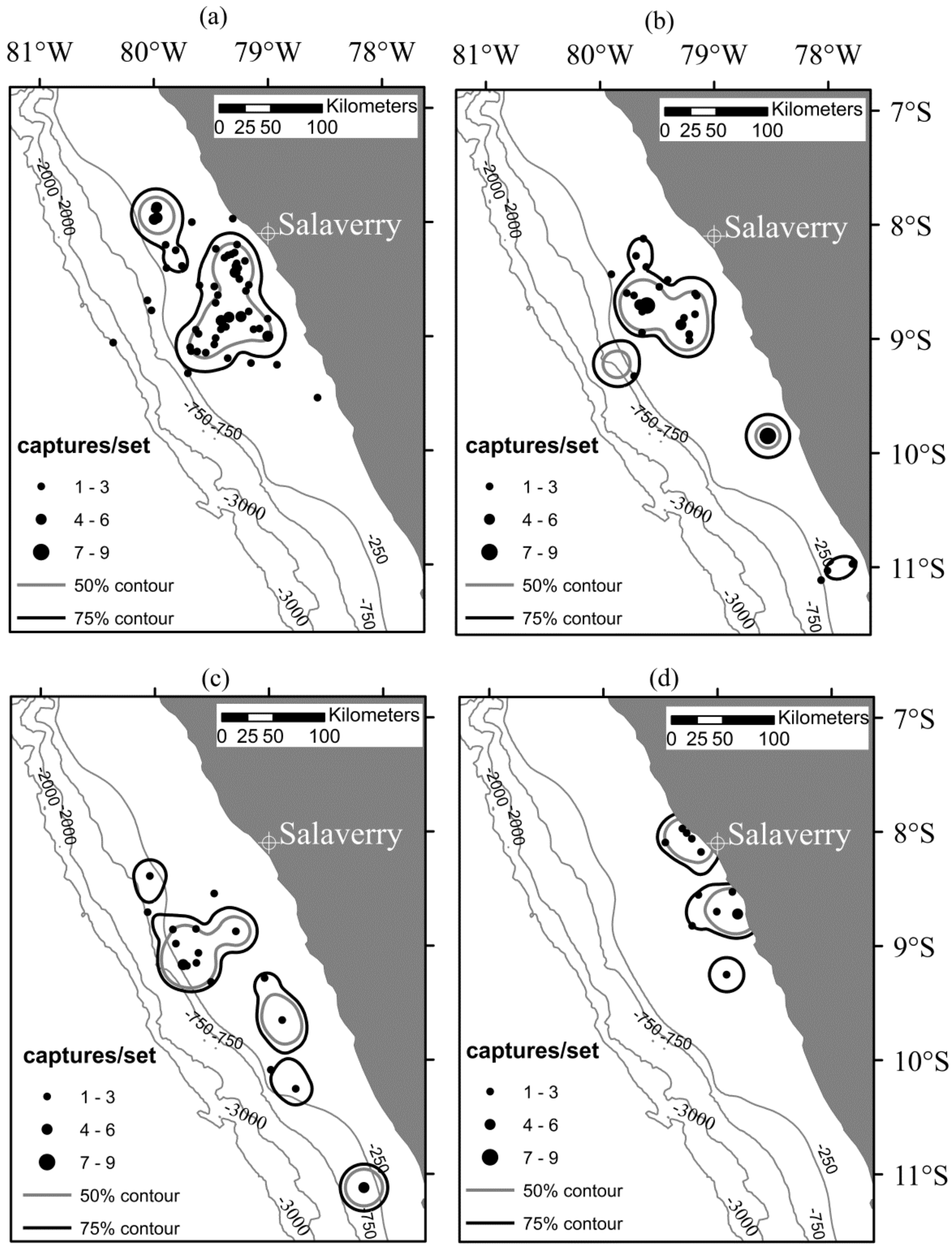
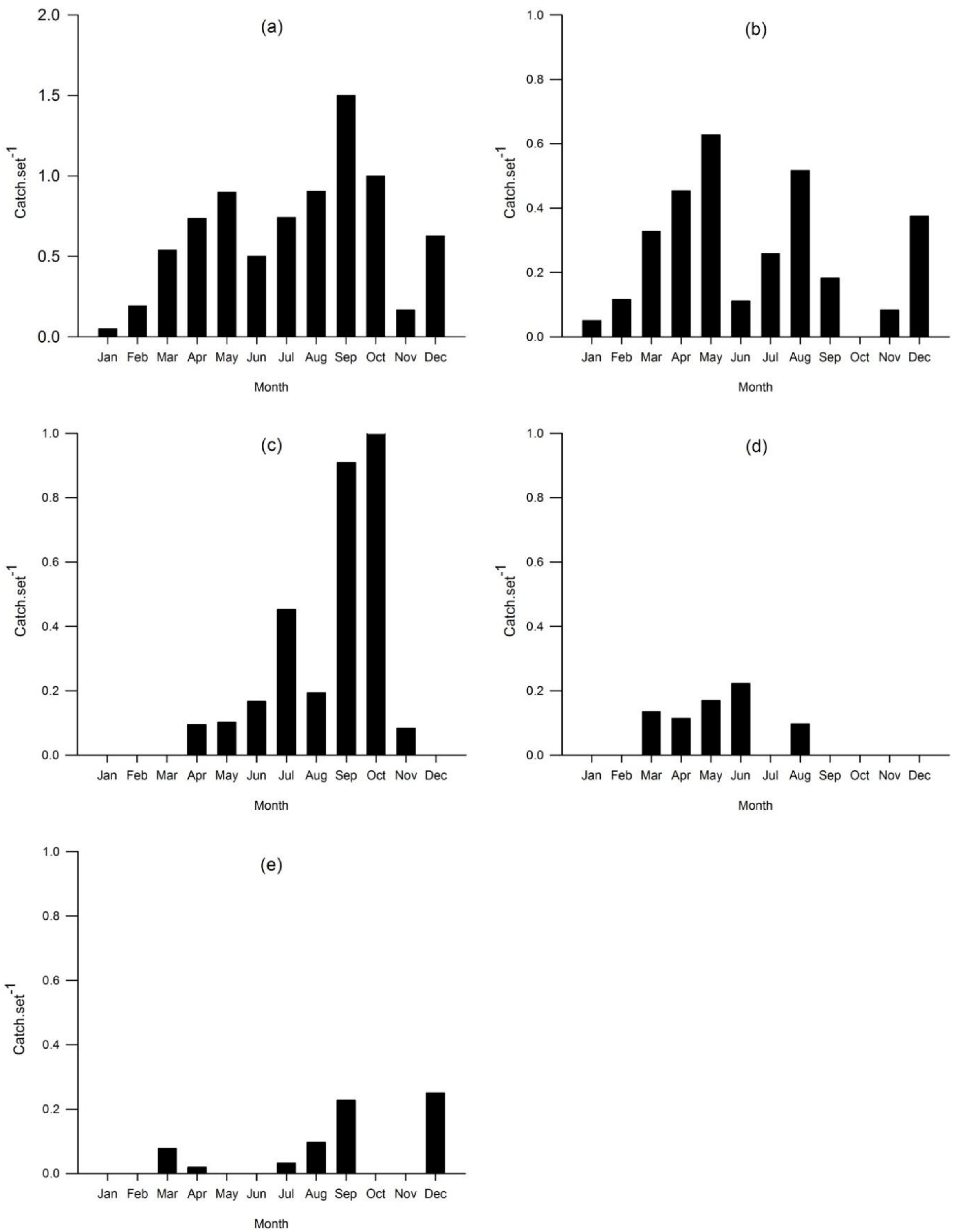


Figure S2. Mean monthly gillnet bycatch.set^{-1} of (a) total small cetaceans, (b) *Delphinus* spp., (c) *L. obscurus*, (d) *T. truncatus,* and (e) *P. spinipinnis*. Data are pooled by month for the study period of March 2005 - July 2007.



Chapter IV: Using pingers to reduce small cetacean bycatch in the small-scale driftnet fishery in Peru

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Abstract

There is growing awareness that small-scale fisheries may have large impacts on threatened marine fauna. Bycatch rates of small cetaceans by the Peruvian small scale driftnet fleet are among the highest ever observed and annually result in the deaths of many thousands of animals. We sought to assess the effectiveness of acoustic alarms (pingers) at reducing the incidental capture of dolphins and porpoises by this fleet. Forty three experimental trips (156 fishing sets) and 47 control trips (195 fishing sets) were observed from April 2009 to August 2011 from Salaverry Port, northern Peru. Twenty-two percent of control sets captured small cetaceans (67 animals) while 16% of experimental sets had captures of small cetaceans (33 animals). The bycatch rate of experimental sets was 0.50 animals $(km^2h)^{-1}$, while for control sets the rate was 0.80 animals $(km^2h)^{-1}$. This 37% reduction in bycatch rate suggests that pingers may be effective at reducing the bycatch of small cetaceans in this fishery. Catch rates of target shark and ray species were unchanged. The potential impacts of our findings to small cetacean conservation in the southeastern Pacific Ocean are substantial given the vast size of this fishery and its current levels of small cetacean bycatch (> 10 000 animals annually). Challenges to large scale implementation of pingers remain to be overcome, including costs. The continued harpooning of dolphins for use as bait will also need to be addressed in order for the potential dramatic reductions in dolphin and porpoise bycatch and mortality to be achievable.

Introduction

Small scale fisheries are globally important as food providers and sources of employment in many coastal communities, particularly in the developing world (Béné 2006; Berkes et al. 2001; Chuenpagdee et al. 2006; McGoodwin 2001). Unlike industrial fisheries which are more centralized and often subject to more defined management structures, small-scale fisheries are frequently characterised by political and economic marginalisation and are often subject to minimal management and enforcement mechanisms (Berkes et al. 2001; Dutton and Squires 2008; Jacquet and Pauly 2008; McGoodwin 2001). A growing list of recent studies have made clear that small-scale fisheries can have significant levels of incidental catch, or "bycatch", of marine fauna (e.g. Alfaro-Shigueto et al. 2011; Jaramillo-Legorreta et al. 2007; Mangel et al. 2010; Moreno et al. 2006; Peckham et al. 2007). Attempts to address bycatch in these fisheries, through introduction of mitigation measures or time-area closures for example, are, however, highly challenging (Campbell and Cornwelles 2008; Lewison et al. 2004a; Soykan et al. 2008).

The bycatch of small cetaceans has been reported in many fisheries worldwide (Jefferson and Curry 1994; Perrin et al. 1994; Read et al. 2006; Reeves et al. 2003). Small-scale fisheries are likely to contribute significantly to this take (Read 2008; Read et al. 2006). Gillnet fisheries in particular are widely seen as one largest global sources of small cetacean mortality (Dawson and Slooten 2005; Jefferson and Curry 1994; Read 2008; Read et al. 2006).

The use of acoustic alarms is one of the few potential solutions to gillnet bycatch that have been identified (Read 2008). Acoustic alarms, or "pingers", are small battery powered devices attached at intervals along a net and which emit a repeated signal audible to small cetaceans. Pingers have been shown to be successful at reducing the bycatch of multiple cetacean species (e.g. Barlow and Cameron 2003; Bordino et al. 2002; Carretta et al. 2008; Kraus et al. 1997; Palka et al. 2008) and their use is now a required bycatch mitigation measure in several commercial net fisheries in the United States and Europe (European Commission 2004, NOAA & NMFS 1997, 1998).

In Peru, fishery interactions with small cetaceans have been reported since the 1960s (Clarke 1962; Read et al. 1988; Van Waerebeek and Reyes 1990). The majority of interactions were of small cetaceans caught in the small-scale gillnet fleet, and the annual take was estimated to range from 10 000 to 20 000 animals (Read et al. 1988; Van Waerebeek and Reyes 1990, 1994b).

Ministerial decrees in 1990 and 1994 (Decree No. 321-94-PE), were followed by a national law in 1996 banning the capture and trade in small cetaceans (Ley No. 26585, 9 April 1996). Subsequent monitoring indicated that small cetaceans were still killed in Peru's small scale fisheries but reporting was limited (Van Waerebeek et al. 2002a; Van Waerebeek et al. 1997b). Using onboard observers on driftnet vessels from the northern Peru port of Salaverry, Mangel et al. (2010) reported that small cetacean bycatch was still common and suggested that capture rates may still be at levels similar to before the national ban in 1996. This work further noted that the majority of small cetacean bycatch was unwanted and discarded, suggesting to the authors an opportunity to introduce small cetacean bycatch mitigation measures, such as pingers, to the fishery. The purpose of the current study was to assess the effect of pingers on small cetacean bycatch and target catch under true fishery conditions within the Peruvian small-scale driftnet fishery.

Methods

The fishery

Small-scale or "artisanal" vessels are defined according to Peruvian fishery regulations as those boats with a maximum of 32.6 m^3 of storage capacity, less than 15 m in length, and principally based on the use of manual work during fishing operations (Ley General de Pesca, 2001). From April 2009 to August 2011 small-scale driftnet fishing trips were monitored out of the port of Salaverry (8°14'S, 78°59'W) in northern Peru. Vessels in this fishery set multifilament net at the ocean surface during the late afternoon and recover the net the following morning, after a soak time of approximately 12 hours. This fleet operates almost exclusively over Peru's continental shelf and targets shark and ray species (primarily smooth hammerhead (*Sphyrna zygaena*), blue (*Prionace glauca*), short-fin mako (*Isurus oxyrinchus*), and thresher (*Alopias vulpinus*) sharks, and eagle rays (*Myliobatis* spp.)), although other species are captured incidentally, including sea turtles (Alfaro-Shigueto et al. 2011), seabirds (Awkerman et al. 2006), swordfish (*Xiphias gladius*), dolphinfish (*Coryphaena hippurus*) and manta rays (*Manta birostris*). Detailed descriptions of this fishery and its capture species can be found in Mangel et al. (2010) and Alfaro-Shigueto et al. (2010) and Table 1.

Experimental design

Monitored vessels undertook normal fishing operations and did not deviate from their normal procedures or fishing locations for the purposes of the experiment. Two fishing captains operating from six different vessels took part and were voluntary participants in the project. Control and experimental sets occurred in all months of the study except August (2009, 2010) when vessels underwent annual maintenance. Vessels either set their nets continuously without pingers (control sets) or with pingers (experimental sets) or alternated between control and experimental sets throughout the course of a fishing trip. There was minimal variation in net characteristics (mesh size, net length and area), set time and duration, and set location between control and experimental sets (Table 1).

Dukane Netmark 1000 pingers were used in the study. These pingers have a fundamental frequency of 10 kHz and emit a 300 msec tone every 4 seconds with a source level of 130 dB (re: 1

μPa @ 1 m). For experimental sets, pingers were attached to the net floatline and placed at a 200 m interval along the entire length of the net. Pingers were checked before and after each deployment and any failed units were replaced. Data from those sets with failed pingers were not included in the analysis.

Data collection

Onboard observers monitored all control and experimental fishing sets. These observers were trained how to maintain and deploy the pingers and how to monitor relevant aspects of the fishery operation as well as target catch and small cetacean bycatch. Observers also recorded the provisioning costs for each trip and the gross profits received when the catch was landed and sold. Variables included in onboard observer monitoring were the date, time and location of sets as well as the primary gear dimensions and characteristics. Small cetacean blubber is frequently used as bait in this fishery (either from harpooned or bycaught animals) and this practice was also monitored. For each fishing set, all target catch and bycatch was counted and identified to the species level whenever possible. Bottlenose dolphins (*Tursiops truncatus*) were not differentiated between offshore or inshore stock. Neither common dolphins (*Delphinus* spp.) nor pilot whales (*Globicephala* spp.) were identified to the species level given uncertainties in at-sea identification (it is likely, however, that the vast majority of common dolphin interactions were with *D. capensis* (Mangel et al. 2010; Van Waerebeek 1994)). Observers also monitored whether entangled cetaceans were alive or dead at the time of the haul and the final fate of each animal (released alive, discarded dead, used as bait, used for food).

Data analysis

Pinger effectiveness was assessed using generalised linear mixed models (GLMM). For testing the impact of pinger use on small cetacean bycatch we used Poisson error distributions with fixed effects (control set vs. experiment set) and random effects (trip, year, season) as well as an offset term for fishing effort: offset(log[net area x time]). This offset term was calculated using the onboard observer data on each set's net dimensions (km^2) and duration (h) and was therefore estimated as km^2h. Use of the log-offset allows the intercept parameters estimated by GLMM to be interpreted as catch per unit effort. Small cetacean bycatch was not over-dispersed and model checks confirmed that the Poisson error structure was valid. Seasons were defined as follows:

quarter 1 = January to March; quarter 2 = April to June; quarter 3 = July to September; quarter 4 = October to December. The dependent variable was the total count of small cetaceans captured during a given fishing set.

We also examined the impact of pinger use on the target catch of sharks and rays. These tests also employed GLMMs and were structured similar to the tests for small cetaceans described above. However, as shark and ray catch data were highly over-dispersed, we included an additional individual level random effect term which served to fit the extra-Poisson variation as a normally distributed error around the intercept (Elston et al. 2001). The dependent variable here was the total count of sharks and rays captured during a given fishing set.

Using the GLMMs we were able to calculate the small cetacean and target catch for control and experimental sets. This was accomplished by back-transforming the intercepts of the control and treatment groups to derive the catch per unit effort which is presented as catch $(km^2h)^{-1}$ (Table 3).

All GLMMs were fitted using the lme4 package (http://lme4.r-forge.r-project.org) for the open source statistical modeling program R 2.13.1 (R Development Core Team 2011). Spatial analyses and maps were prepared using ESRI ArcMap 9.2 and Hawth's Tools Extension (www.spatialecology.com/htools). Bathymetric values were obtained from the Global Bathymetric Chart of the Oceans (GEBCO, www.gebco.net; IOC 2003). Descriptive statistics are presented as mean ± standard deviation (SD) unless specified otherwise. Trip costs and profits are presented in US dollars.

Results

Fishing effort

Over the 29 months of the study we observed 195 control sets over 47 trips and 156 experimental sets over 43 trips (Table 1, Fig. 1). Small cetacean blubber was used as bait in 24% of control sets and 31% of experimental sets. The average cost to provision trips was \$1020 ± \$669 (Range: \$220 to \$4405, n = 52) and average gross profits were \$2195 ± \$1594 (Range: \$0 to \$7401). Net profits were \$1176 ± \$1468 (Range: -\$2276 to \$6094) with ten trips (19.2%) operating at a loss.

Small cetacean bycatch

Five species of small cetaceans were observed captured including common dolphins, dusky dolphins (*Lagenorhynchus obscurus*), bottlenose dolphins, Burmesiter's porpoises (*Phocoena spinipinnis*), and pilot whales. Each of these species was caught in both control and experimental sets, except pilot whales which were only caught on one occasion in a control set. Forty-five percent of all small cetacean bycatch was common dolphins. All small cetaceans died as a result of their entanglement. The larger bycaught animals, including bottlenose dolphins and pilot whales were typically discarded, while the majority of dusky dolphins were butchered for use as bait and the majority of Burmeister's porpoises were butchered for use as food (Table 2).

In addition to bycatch events, 23 common dolphins and 2 dusky dolphins were observed harpooned during the study for use as bait on subsequent sets. This typically occurred as vessels were traveling to the fishing ground prior to the first fishing set. Harpooning occurred on 10 trips (23%) and ranged from 1 to 4 animals per event. Twenty common dolphins were harpooned during 7 trips which used pingers while 2 dusky dolphins and 3 common dolphins were harpooned during 3 control trips.

Pinger effectiveness

A total of 100 small cetaceans were observed captured, 67 during control sets and 33 during experimental sets (Table 2, S1, Fig. 2). Twenty-two percent of control sets (43 sets) and 16% of experimental sets (25 sets) had small cetacean bycatch. Control sets had a maximum catch of four animals in a given set while experimental sets had a maximum catch of three animals. Sets using pingers had a 37.2% lower bycatch rate of small cetaceans and this difference was statistically significant (GLMM, χ^2_1 = 4.0158, *p* = 0.0450; Table 3). The bycatch rate declined from 0.7980 $(km^2h)^{-1}$ (range ± 1 standard error [SE]: 0.6781 to 0.9393) for control sets to 0.5015 $(km^2h)^{-1}$ (range ± SE: 0.4067 to 0.6185) for experimental sets. There were declines in bycatch rates of each small cetacean species but none of these declines were statistically significant (Table 3, S1). Variable "trip" dominated variation among levels of the random effects while variance due to bait use, year and season was negligible.

Control sets had a catch rate of 18.6 sharks $(km^2h)^{-1}$ (range ± SE: 14.0 to 24.7) while experimental sets had a catch of 26.3 sharks $(km^2h)^{-1}$ (range ± SE: 19.7 to 35.0) but this difference was not statistically significant (GLMM, χ^2_1 = 2.9157, *p* = 0.088**,** Table 3, Fig. S1). Likewise, there was no statistical difference in the catch rates of rays between control sets (GLMM, χ^2_1 = 0.0534, *p* = 0.82) and experimental sets (Table 3, Fig. S1), with catch rates of 0.0014 $(km^2h)^{-1}$ and 0.0018 $(km^2h)^{-1}$, respectively.

Discussion

Given the nature of small scale fisheries (e.g. minimal management or enforcement, economic and political marginalization), efforts to identify, test and implement bycatch mitigation measures have proven challenging (Campbell and Cornwelles 2008; Soykan et al. 2008). Here, in the first study of its kind in the southeastern Pacific Ocean, we tested pinger effectiveness and have shown that they reduced small cetacean bycatch in the Peruvian small-scale driftnet fishery. As was observed in the California drift gillnet fishery for swordfish and sharks, the reduction was most pronounced when assessing total small cetacean bycatch (Barlow and Cameron 2003). Given the modest sample size and relative rarity of bycatch events, the observed declines in bycatch at the species level were not statistically significant. Moreover, while use of pingers did reduce the bycatch rate by 37%, it did not eliminate it. The greatest decline in bycatch rate was observed for common dolphins (44% decline) and this is similar to observations in the California drift gillnet fishery (Barlow and Cameron 2003; Carretta and Barlow 2011).

Pinger effectiveness

Pinger spacing varies among published studies (e.g. Barlow and Cameron 2003; Bordino et al. 2002; Gazo et al. 2008; Gönener and Bilgin 2009), while other studies indicate that dolphins and porpoises alter their behavior or distribution in the presence of pingers (Berrow et al. 2008), at distances beyond 500m (Carlstrom et al. 2009). In the present study we spaced pingers at 200m in an attempt to strike a balance between pinger effectiveness and a realistic, cost effective implementation strategy in this small-scale fishery. The decrease in small cetacean bycatch rate that we observed (37%) was less than has been observed in other studies, where reductions ranged from approximately 50% to 90% (Barlow and Cameron 2003; Bordino et al. 2002; Carretta and Barlow 2011; Carretta et al. 2008; Kraus et al. 1997). Reasons for differences between studies could be many and include variations in fishery characteristics and net type, target and bycatch species, abundance and group size, pinger specifications, as well as varying methods used to calculate bycatch rates. Further reductions in the small cetacean bycatch rate in Peru may, however, be possible if pinger spacing were reduced as there is some evidence that bycatch rates decrease as the number of pingers used increased (Barlow and Cameron 2003; Trippel et al. 1999).

It is interesting to note also that there was no observed bycatch of pinnipeds during the study although South American sea lions (*Otaria flavescens*) and fur seals (*Arctocephalus australis*) are common on the fishing grounds and are known to depredate catch from fishing nets. Concerns have been raised that the sound emitted by pingers could increase pinniped depredation by alerting them to the presence of nets, commonly referred to as the “dinner bell” effect (Dawson 1991). While systematic monitoring of pinniped depredation was beyond the scope of this study, it should be monitored going forward as the effectiveness of pingers could be undermined if catch values declined as a result of increased damage from pinnipeds and other species (Bordino et al. 2002; Carretta and Barlow 2011; Kraus et al. 1997).

There was no statistically significant difference in catch rates between control and experimental sets of sharks and rays, the primary target species in this fishery. This finding is in line with other pinger trials which showed either no impact on target catch (Bordino et al. 2002; Carlstrom et al. 2002; Gazo et al. 2008; Gönener and Bilgin 2009; Kraus et al. 1997; Trippel et al. 1999) or an improved target catch (Buscaino et al. 2009). The lack of an impact in the present study is not unexpected as shark hearing is typically in the 40 to 800 Hz range (Myrberg 2001), well below the 10 KHz fundamental frequency of the Dukane pingers. Moreover, while sharks and small cetaceans are typically considered as having a predator-prey relationship, given the species and small sizes of sharks typically captured in the fishery (approx. 1 m, ProDelphinus unpublished data), it may be more accurate to consider them to be primarily competitors for prey (Heithaus 2001).

Regional significance and barriers to implementation

The significance of the declines in bycatch rate associated with pinger use become clear when one considers the potential regional level impacts. Given the catch rates reported here and the level of annual fishing effort for the port of Salaverry (Mangel et al. 2010), if all gillnet vessels in the port used pingers, one could expect a reduction in small cetacean bycatch on the order of 500 animals a year. As the Salaverry gillnet fleet is ca. 2% of gillnet fishing effort in Peru (Alfaro-Shigueto et al. 2010; Escudero 1997; Estrella et al. 1999; Estrella et al. 2000; Estrella and Swartzman 2010; Mangel et al. 2010), use of pingers by gillnet vessels throughout Peru has the potential to reduce small cetacean mortalities by many 1000s of animals per annum.

Barriers to implementation do, however, still exist. The current unit cost of pingers is approximately $80. To equip an average 2 km length net in this fishery would require an investment of $720 for 200m spacing or $1440 for 100m spacing. As we report here, this is about equal to the average net profit of a fishing trip. Modern pinger designs are quite robust and typically have a multi-year battery life. These characteristics would be particularly important for implementation in a small scale fleet where vessels are numerous and distributed among many, often remote, locations. But there would still be additional costs associated with maintaining pingers (e.g. battery failure, pinger damage, loss). Whether vessel owners in this fishery would be willing to accept the costs associated with using pingers in a management environmental in which enforcement is minimal is open to question. As Alfaro-Shigueto et al. (2010) note, gillnet fisheries in Peru can be thought of as an entry-level or “gateway” fishery due to their relatively low costs and profits in comparison to other fishing gears such as longlines. Any efforts to promote pinger use should therefore also stress the potential benefits to work efficiency. Reduced net damage associated with pinger use has been reported (Buscaino et al. 2009; Culik et al. 2001; Gazo et al. 2008). Moreover, similar to the finding in Mangel et al. (2010), we found that approximately half of all dolphin and porpoise bycatch was discarded, a potential source of further net damage and lost time and effort associated with disentanglement.

The use of small cetaceans for bait is widespread globally (Avila et al. 2008; Dolar 1994; Goodall et al. 1994; Lescrauwaet and Gibbons 1994; Mangel et al. 2010; Mora-Pinto et al. 1995; Van Waerebeek et al. 1997b; Zavala-Gonzalez et al. 1994). We observed harpooning of dolphins for use as bait on nearly a quarter of observed trips, and this was particularly prevalent on trips using pingers. While this may be an anecdotal finding it could also reflect the perceptions of participating fishermen of the effectiveness of pingers at reducing bycatch thus necessitating harpooning in order to obtain sufficient dolphin blubber for use as bait for the trip. If harpooning were to increase as a result of using pingers it could largely offset much of the gain made through their use. Under the current management structure, the regular use of dolphins for bait to catch sharks and rays will likely continue.

Conclusions and future directions

We have shown that pingers were effective at reducing small cetacean bycatch in the Peruvian small-scale driftnet shark fishery. The potential benefits of this finding to small cetacean conservation in the southeastern Pacific Ocean are substantial given the vast size of this fishery and its current levels of small cetacean bycatch (Alfaro-Shigueto et al. 2010; Mangel et al. 2010). Further collaborative research with fishermen and vessel owners in the Peruvian gillnet fishery are currently underway and will continue to monitor pinger effectiveness and their impacts on small cetaceans and the fishery (e.g. target catch rates, pinniped depredation). Challenges to large scale implementation do, however, still remain, including the costs associated with pingers and their maintenance as well as the continued prevalence of harpooning of dolphins for use as bait. These issues will have to be addressed in order to expand pinger use to the national level and in order for the potential dramatic reductions in dolphin and porpoise bycatch and mortality to be achievable.

Acknowledgements

We warmly thank the participating fishermen and field observers for their generous collaboration. ProDelphinus biologists, Natalia Ortiz and Graciela McEvoy provided superb field work. Project funding and equipment was provided by Cetacean Society International, Chelonia Ltd., Cleveland Metroparks Zoo, Comisión Permanente del Pacífico Sur, Darwin Initiative, Oak Foundation through Duke University, Rufford Foundation, Whale and Dolphin Conservation Society, as well as Dr. Andy Read of Duke University and Dr. Per Berggren of Newcastle University. JCM and JAS are Overseas Research and Students Awards Scheme (ORSAS) and University of Exeter scholarship awardees, respectively. MJW is funded by the Peninsula Research Institute for Marine Renewable Energy (PRIMaRE). DH is supported by the Natural Environment Research Council and the European Union. BJG receives funding from the Darwin Initiative, the Natural Environment Research Council and the European Union.

Table 1. Gear characteristics and fishing effort for control (no pinger) and experimental (pinger) fishing sets observed during the study, April 2009 to August 2011. Mean ± SD (Range).

	Treatment	
Category	No pinger	Pinger
Sets	195	156
Small cetacean bycatch	67	33
% sets with small cetacean bycatch	22	16
% sets using bait	24	31
Stretched mesh size (in)	7.5, 8, 10	7.5, 8, 10
Set duration (h)	12.7 ± 2.4 (3.6 to 19.2)	12.7 ± 2.6 (2.3 to 16.4)
Total soak time (h)	2477.75	1987.65
Net length (km)	2.02 ± 0.33 (1.81 to 2.96)	1.98 ± 0.36 (1.64 to 2.96)
Net area (km^2)	0.027 ± 0.005 (0.023 to 0.041)	0.027 ± 0.005 (0.023 to 0.041)
Net area time (km^2h)	0.343 ± 0.090 (0.100 to 0.751)	0.347 ± 0.093 (0.053 to 0.643)

Table 2. Species composition of small cetacean bycatch for control (no pinger) and experimental (pinger) fishing sets. Also shown is the final use of the carcasses, presented as percentage, by species, within each of the given categories. "Discarded" refers to animals recovered dead and discarded at sea. "Bait" refers to animals recovered dead and then butchered for use as bait during subsequent fishing sets. "Food" refers to animals recovered dead and then butchered for food either for use during the trip or for home consumption.

	Treatment			Final use (%)			
Species	Total n	No pinger	Pinger	discarded	bait	food	(n)
Delphinus spp.	45	33	12	41	44	12	34
L. obscurus	20	11	9	32	58	0	19
T. truncatus	25	16	9	67	33	0	18
P. spinipinnis	8	5	3	43	0	57	7
Globicephala spp.	2	2	0	100	0	0	2
TOTAL	100	67	33				

Table 3. Catch rates of bycatch (dolphins and porpoises) and target catch (sharks and rays) for all observed control (no pinger) and experimental (pinger) sets, April 2009 to August 2011. Catch rates are presented as catch $(km^2h)^{-1}$ and are derived from the generalized linear mixed models used to test pinger effectiveness. Values shows are the mean and range ± 1 standard error (SE). Also shown is the percent change in catch rate between control and experimental sets and associated p-values.

		No Pinger		Pinger		%	
Category	Species	Mean	± 1 SE	Mean	± 1 SE	change	*p* =
Dolphins & porpoises	TOTAL	0.7980	(0.6781 – 0.9392)	0.5015	(0.4067 – 0.6185)	- 37.2	0.045
	Delphinus spp.	0.2894	(0.2253 – 0.3718)	0.1602	(0.1137 – 0.2256)	-44.6	0.093
	L. obscurus	0.0481	(0.0290 – 0.0797)	0.0431	(0.0254 – 0.0733)	-10.4	0.827
	T. truncatus	0.0507	(0.0302 – 0.0850)	0.0307	(0.0167 – 0.0563)	-39.4	0.360
	P. spinipinnis	0.0008	(0.0001 – 0.0086)	0.0002	(0.0000 – 0.0043)	-75.0	0.379
	Globicephala spp.	0.0001	(0.0000 – 0.0919)	0.0000	na	-100.0	0.692
Sharks & rays	Sharks	18.6	(14.0 – 24.7)	26.3	(19.7 – 35.0)	+29.3	0.088
	Rays	0.0014	(0.0003 – 0.0067)	0.0018	(0.0004 – 0.0090)	+22.2	0.817

Figure 1. Locations of control (filled circles) and experimental (open circles) fishing sets observed over the 29 months of the study (April 2009 to August 2011). Fishing vessels participating in the study were based in the port of Salaverry.

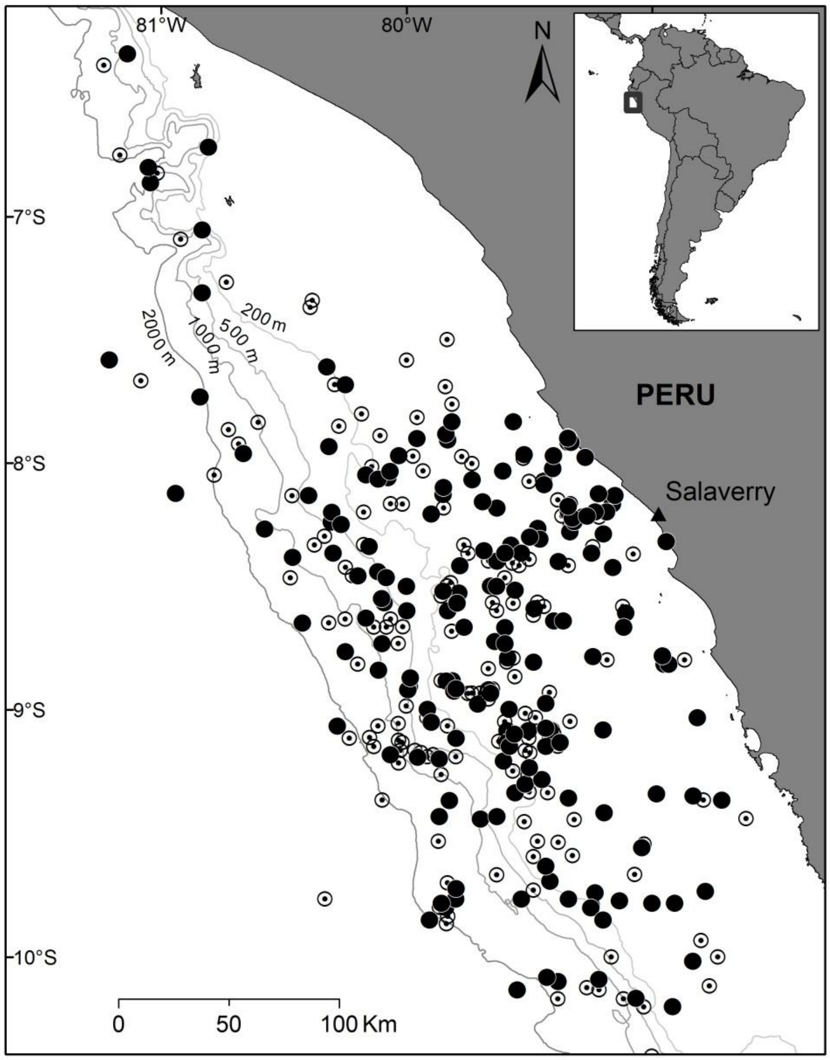


Figure 2. Locations and quantities of small cetacean bycatch in control (panel a) and experimental sets (panel b). The shaded area is the minimum convex polygon of fishing sets for each.

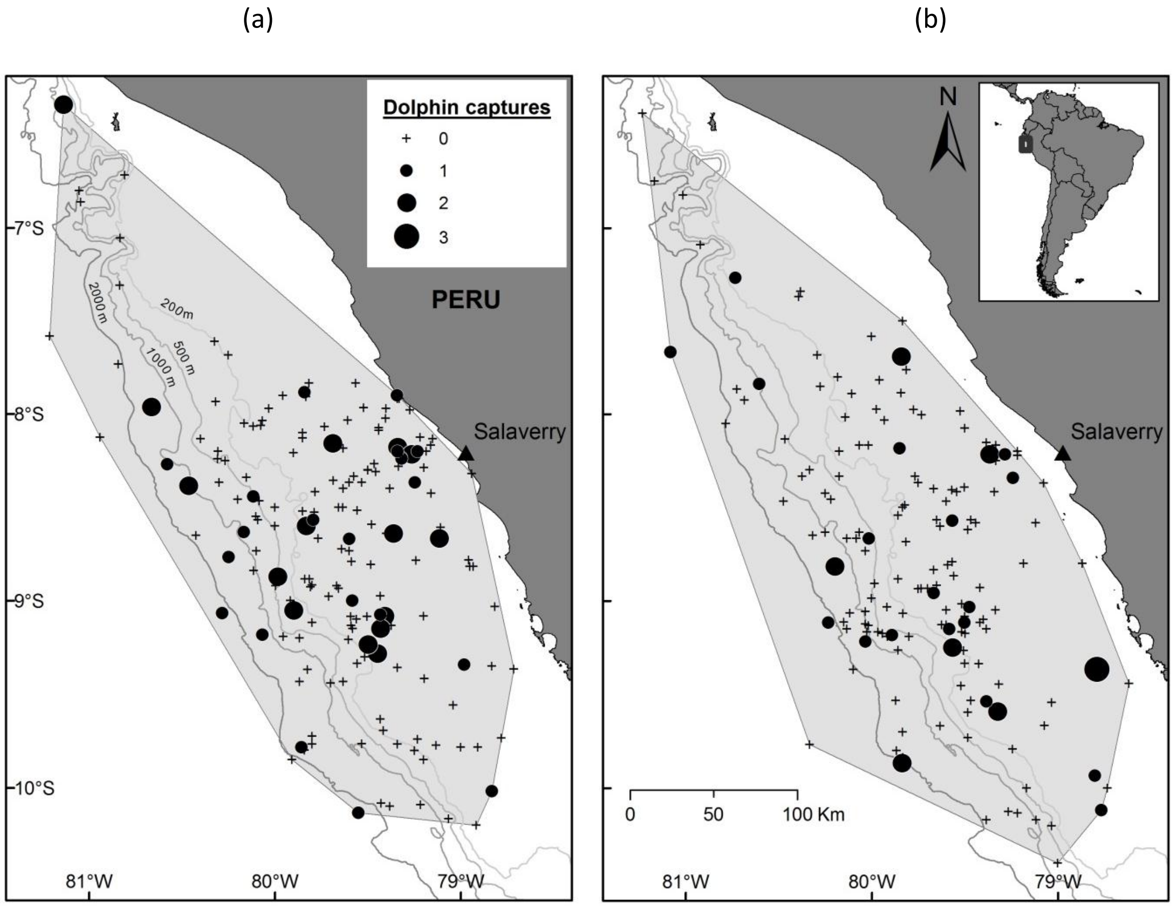


Table S1. Per set catch rates for all small cetacean bycatch species by control (no pinger) and experiment (pinger) treatment groups.

	Catch per set									
	Control					Experiment				
Species	0	1	2	3	4	0	1	2	3	4
Total bycatch	152	23	17	2	1	131	18	6	1	0
Delphinus spp.	171	16	7	1	0	145	9	2	0	0
L. obscurus	187	6	2	0	0	150	4	1	1	0
T. truncatus	186	4	3	1	1	149	6	1	0	0
P. spinipinnis	191	3	1	0	0	154	1	1	0	0
Globicephala spp.	194	0	1	0	0	156	0	0	0	0

Figure S1. Locations and quantities of shark and ray catch in control (panel a) and experimental sets (panel b).

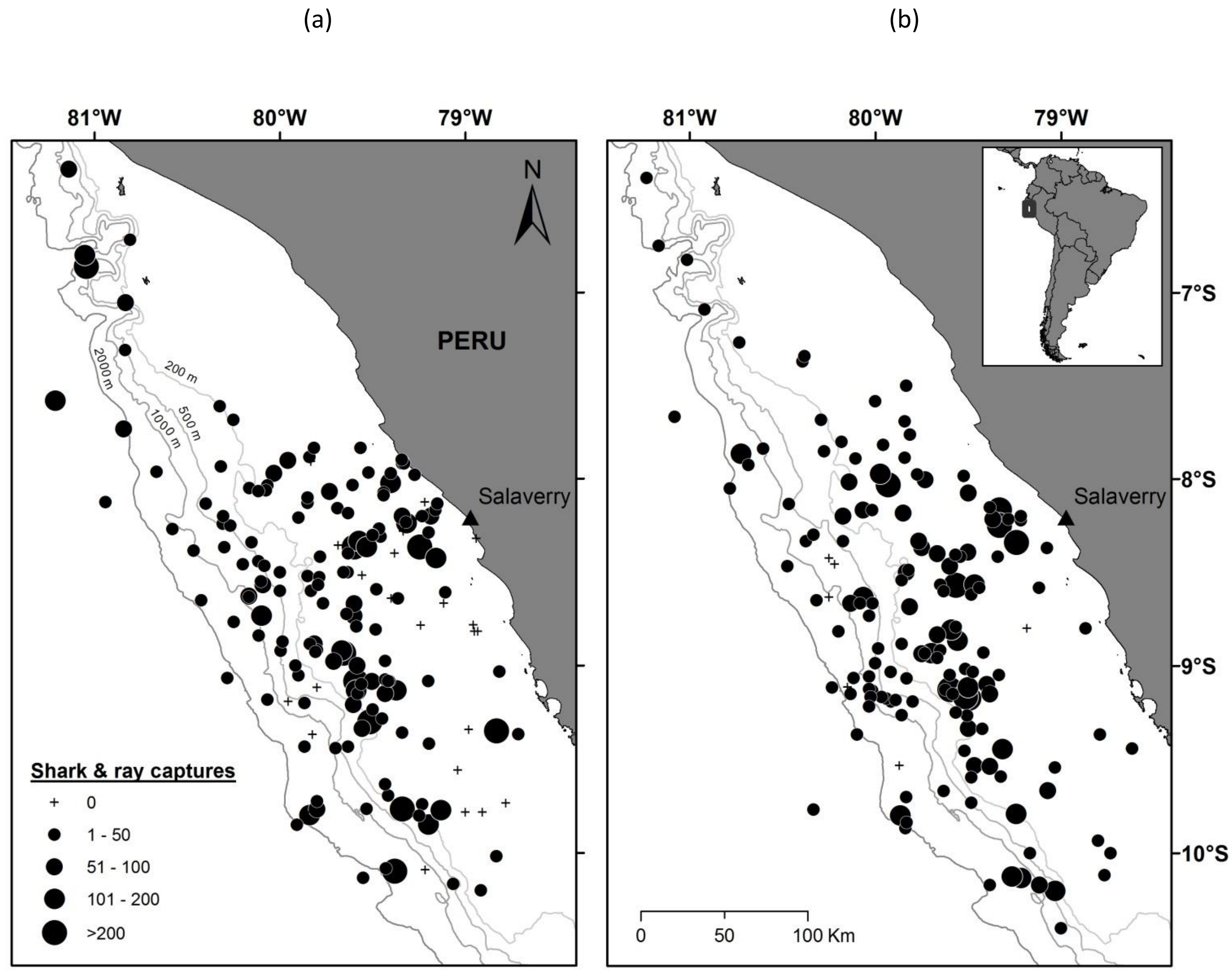
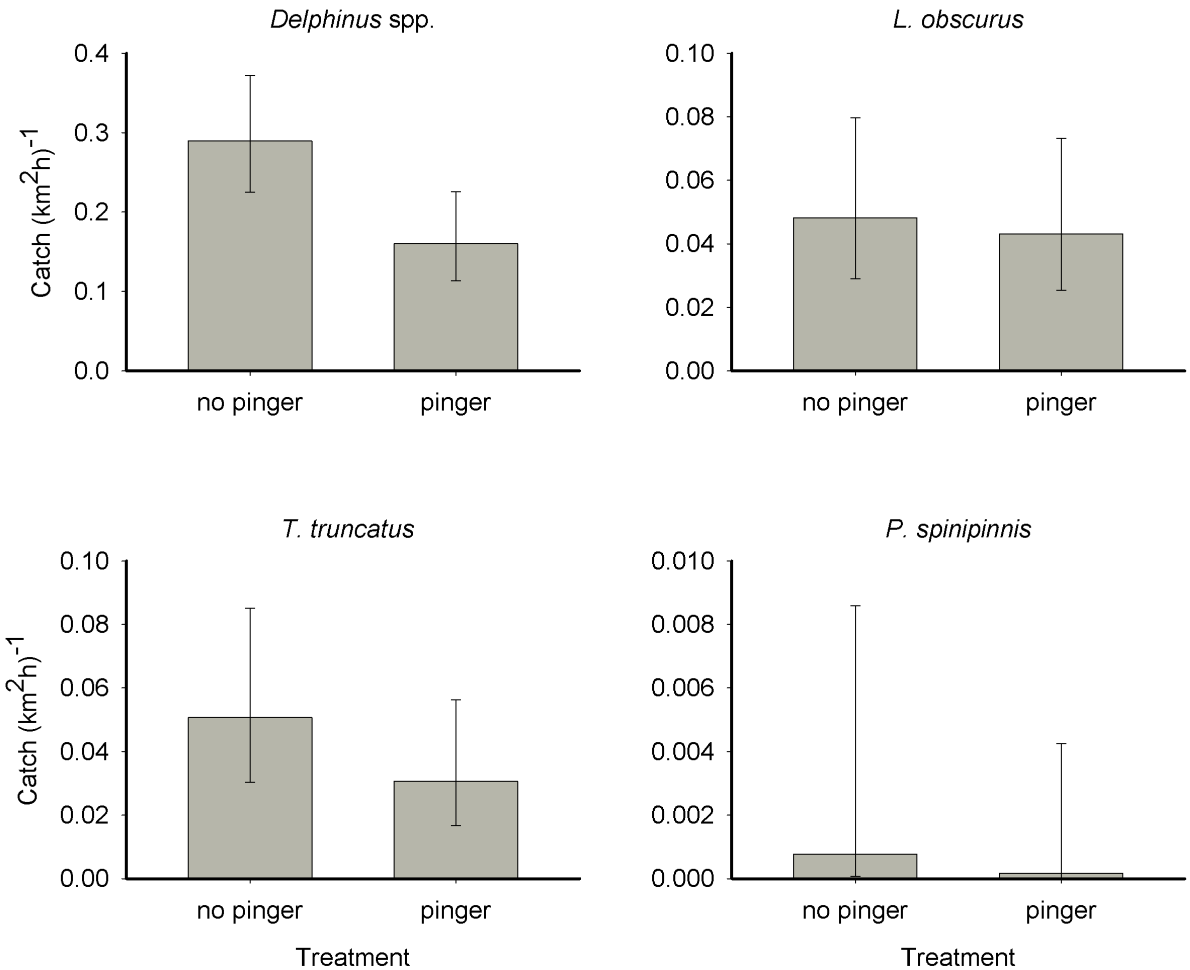


Figure S2. Bar-plots with standard error bars comparing the bycatch rates (catch set^{-1}) for the main four small cetacean species for control (no pinger) and experimental sets (pinger).



Chapter V: Onboard observer data suggest that small-scale fisheries are a major potential threat to seabirds in the southeastern Pacific

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Manuscript in preparation

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Abstract

Fisheries bycatch is the primary anthropogenic threat to many seabird species but its incidence in small-scale fisheries is little studied. We detail the first at-sea monitoring of interactions of seabirds with small-scale longline and gillnet fisheries in Ecuador and Peru, southeastern Pacific Ocean. A total of 115 seabirds were captured during monitoring of 450 longline trips (4 surface, 1 midwater, 2 dermesal) and 213 surface driftnet trips (2 fisheries). Five of the nine monitored fisheries had seabird bycatch (surface longlines for tuna, dolphinfish and sharks, demersal longlines for hake, driftnets for sharks). Bycatch included at least 16 species, including critically endangered waved albatrosses, the endangered black-browed albatrosses, vulnerable Chatham albatrosses, pink-footed shearwaters, Humboldt penguins, black petrels, and white-chinned petrels and near threatened Buller's albatrosses, guanay cormorants, Inca terns and sooty shearwaters. Bycatch rates for the Ecuadorian longline fisheries (Hake: Mean ± SD; 0.112 ± 0.224 seabirds 1000 hooks^{-1} and; Tuna: 0.237 ± 0.339 seabirds 1000 hooks^{-1}) were approximately 3 to 10 times higher than those for Peruvian longline fisheries. Based upon catch rates reported here and known fleet sizes, annual seabird catch rates likely range from ca. 1000 (demersal longline for hake) to over 10 000 (surface driftnets for sharks and rays). Given the vast sizes and distributions of these fisheries and the observed catch rates of seabirds, many of which are globally threatened, we conclude that there is an urgent need for more intensive, regional efforts toward better understanding and mitigating these interactions.

Introduction

Fisheries bycatch is widely recognized as the primary anthropogenic threat to many seabird species (Brothers 1991; Brothers et al. 1999; Bull 2007; Gilman et al. 2005; Gilman et al. 2008b; Lewison et al. 2004a; Tasker et al. 2000). Attention has largely focused on the seabird bycatch caused by large-scale fisheries, primarily by pelagic longlines (Anderson et al. 2011; Brothers et al. 1999; Bull 2007) but commercial trawl (Gonzalez-Zevallos et al. 2007; Melvin et al. 1999; Sullivan et al. 2006b; Weimerskirch et al. 2000; Zador et al. 2008) and net fisheries (Benjamins et al. 2008; Cardoso et al. 2011; Carretta et al. 2004; DeGange and Day 1991; Piatt and Netteship 1987; Uhlmann et al. 2005; Zydelis et al. 2009) are also recognized as globally significant sources of mortality.

There is now growing recognition that small-scale longline and net fisheries may also have significant impacts on many marine vertebrates including sea turtles (Alfaro-Shigueto et al. 2011; Gilman et al. 2010; Peckham et al. 2007; Peckham et al. 2008), marine mammals (Mangel et al. 2010; Moore et al. 2010; Read 2008) and seabirds (Bugoni et al. 2008b; Cardoso et al. 2011; Gandini and Frere 2006; Majluf et al. 2002; Moreno et al. 2006) but bycatch in these fisheries remains under-studied (Lewison et al. 2004a; Moore et al. 2010; Soykan et al. 2008). These fisheries, which account for about half of global fish catch (Berkes et al. 2001) and the vast majority of fishers (Moore et al. 2010; Pauly 2006), are vital in many coastal communities, particularly in the developing world, as sources of food and employment (Béné 2006; Chuenpagdee et al. 2006; McGoodwin 2001). Small-scale fisheries are often subject to minimal management or enforcement due in part to the frequent economic and political marginalization of these communities (Jacquet and Pauly 2008; McGoodwin 2001; Salas et al. 2007).

The southeastern Pacific Ocean (SEP) (Fig. 1) is one of four eastern boundary current systems on earth and one of the world's most productive marine ecosystems (Bakun and Weeks 2008; Carr 2002; Taylor et al. 2008). The SEP is also an area of global significance for seabirds, with approximately one-third of all seabird species present in the area (Harrison 1983; Jahncke et al. 2004; Spear and Ainley 2008). Small-scale fisheries along the Pacific coast of South America are extensive and diverse with an estimated 15,000 and 10,000 small-scale fishing vessels in operation, in Ecuador and Peru, respectively (Alfaro-Shigueto et al. 2010; Alvarez 2003; Arriaga and Martinez 2003a). Detailed information on the size and structure of the Ecuador small-scale fleet is limited but it is clear that it has undergone significant growth since the 1980s (Alvarez 2003; CPPS 1986; Pinoargote 2008). The fleet size for Peru represents a 54% increase from 1995 to 2005, with the longline fishery growing by 357% over the same period (Alfaro-Shigueto et al. 2010; Escudero 1997; Estrella 2007). However, at 33%, the largest portion of the fleet is vessels using nets (Estrella 2007).

A number of studies have identified the bycatch of marine vertebrates, including seabirds, sea turtles, marine mammals and sharks, by small-scale longline and net fisheries in the SEP region (Alava 2008; Alfaro-Shigueto et al. 2011; Felix and Samaniego 1994; Gilman et al. 2008a; Majluf et al. 2002; Mangel et al. 2010; Moreno et al. 2006; Simeone et al. 1999; Van Waerebeek et al. 1997b). Documentation of seabird interactions with longline and net fisheries in the SEP is, however, limited and based upon shore-based (Jahncke et al. 2001; Majluf et al. 2002; Simeone et al. 1999) and at-sea observations (Alfaro-Shigueto et al. 2010; Moreno et al. 2006). These authors reported considerable levels of seabird bycatch occurring in both longline and gillnet fisheries impacting a number of threatened species including albatrosses and penguins. Jahncke et al. (2001), based upon surveys with fishermen, estimated from 2370 to 5610 albatrosses per year taken by Peruvian longline fisheries. More recently, Awkerman et al. (2006) reported that incidental and direct take of waved albatrosses (*Phoebastria irrorata*) by Peruvian small-scale fisheries may account for reduced adult survival rates observed at the Galapagos Islands nesting colony and in 2007 the IUCN redlist category of the waved albatross was changed from Vulnerable to Critically Endangered, partly in response to these findings. There are no published reports available on seabird interactions with Ecuadorian small-scale fisheries but there is evidence of bycatch of sea turtle and marine mammals (Alava 2008; Felix and Samaniego 1994; Van Waerebeek et al. 1997b). Based upon the region's importance to seabirds and the sizes of the fishing fleets operating in the area, Anderson et al. (2011) listed the SEP as one of the primary data gaps remaining regarding seabird bycatch in longline fisheries.

The objective of this study was to monitor, describe and quantify, through the use of onboard observers, the bycatch of seabirds by multiple small-scale longline and net fisheries in Ecuador and Peru. Given the critically endangered status of the waved albatross observer effort was centered on fisheries operating in southern Ecuador and northern Peru, areas that overlap with the foraging range of the species (Anderson et al. 1997; Anderson et al. 2003).

Methods

2.1 Onboard observer scheme

Seabird bycatch was monitored by trained onboard observers. All observers were biologists, fisheries technicians or fishermen trained in relevant data collection methods, including seabird identification. Data were gathered on the specific gear used (longline or gillnet), the timing and position (using GPS) of each set, and any bycatch occurring. Bycatch characteristics recorded included timing of event (set, soak, haul), species and quantity of bycatch, location of hooking or entanglement (beak, wing, etc.), and state (live, dead) and fate (released alive, discarded dead, etc.) of the seabird. All observers were equipped with cameras and photographed unusual or unidentifiable captures for later species identification. Observers did not participate in fishing activity and the crews and vessels that hosted observers were voluntary participants in the project. Except in cases when a fishery operated seasonally, observers worked year-round to account for possible seasonal variations in fishing effort or seabird bycatch.

2.1.1 Ecuador

From November 2008 to April 2011 a total of 354 fishing trips (900 sets, 568 fishing days) were monitored for seabird bycatch in Ecuador (Table 1, Fig. 1). Trips monitored were on longline and driftnet fishing vessels operating from the ports of Santa Rosa (02.21°S 80.96°W) and Anconcito (02.33°S 80.89°W). Five longline fleets were monitored: surface longlines for yellowfin tuna (*Thunnus albacares)*, surface longlines for dolphinfish (*Coryphaena hippurus*), midwater longlines for ecolar (*Lepidocybium flavobrunneum*), demersal longlines for south Pacific hake (*Merluccius gayi*), and demersal longlines for Pacific bearded brotula (*Brotula clarkae*). Observers also monitored the driftnet fishery targeting skipjack tuna (*Katsuwonus pelamis*), striped bonito (S*arda orientalis*) and dolphinfish. Detailed specifications of these gear types and observer effort are given in Table S1.

2.1.2 Peru

From May 2005 to May 2011 a total of 309 fishing trips (2156 sets, 1990 fishing days) were monitored for seabird bycatch in Peru (Table 1, Fig. 1). Trips monitored were on longline and driftnet fishing vessels operating from the port of Salaverry (08.23°S, 78.98°W) and longline vessels from the port of Ilo (17.65°S, 71.35°W). Driftnet vessels targeted primarily blue (*Prionace glauca*), shortfin mako (*Isurus oxyrinchus*), thresher (*Alopias vulpinus*), and smooth hammerhead sharks (*Sphyrna zygaena*), and eagle rays (*Myliobatis* spp.). Longline vessels set their gear at the ocean surface and seasonally targeted either blue and mako sharks (March to November) or dolphinfish (December to February). Longline gear configurations between shark and dolphinfish seasons differ (e.g. branchline spacing, leader material, hook size) therefore bycatch results for these fisheries are considered separately. Detailed specifications of these gear types and observer effort are given in Table S1 and in Alfaro-Shigueto (2010).

2.2 Data analysis and mapping

Using methods detailed in Mangel et al. (2010) and Alfaro-Shiugueto et al. (2011), a monthly stratified seabird bycatch per unit effort (CPUE) was calculated for each fishery and is presented in multiple units to allow for comparison with other studies (Table 2, S3). For longline fisheries seabird CPUE is presented per set, per 1000 hooks and per 1000 hooks * fishing set duration in hours ($[1000 \text{ hooks h}]^{-1}$). For gillnet fisheries, CPUE is presented per set and per km^2 of net area * fishing set duration in hours ($[km^2h]^{-1}$). An overall seabird CPUE is presented for each fishery and for species (e.g. waved albatross, white-chinned petrel) or species groups (e.g. albatrosses) of particular concern or for which species level data were sufficient to derive an estimate. Using these calculated CPUE rates and fishing effort characteristics (e.g. average number of sets per trip, average number of hooks per set) and data on port level fishing effort derived from this study (in the case of Ecuador) or national level fishing effort data available reports and publications in the case of Peru (Alfaro-Shigueto et al. 2010; Escudero 1997; Estrella 2007; Estrella et al. 1999; Estrella et al. 2000) we then constructed order-of-magnitude estimates of seabird bycatch for each fishery (Table 3). Descriptive statistics are presented as mean ± standard deviation (SD) unless specified otherwise. Maps were prepared using ESRI ArcMap 9.2 and the Hawth's Tools Extension (www.spatialecology.com) was used to create minimum convex polygons of fishing effort. Bathymetric values were obtained from the Global Bathymetric Chart of the Oceans (GEBCO, www.gebco.net, IOC et al. 2003).

Results

Five of the nine monitored fisheries had observed seabird bycatch (Tables 1, S2, Fig. 1). These included all three fisheries monitored in Peru: driftnets; surface longlines targeting sharks; surface longlines targeting dolphinfish. In Ecuador seabird bycatch was observed in the surface longline fishery targeting yellowfin tuna and the demersal longline fishery targeting South Pacific hake.

3.1 Species composition

A total of 17 seabirds were observed captured in the Ecuadorian dermersal longline fishery for hake and 4 in the surface longline fishery for tuna. The most frequent bycatch species in the Ecuadorian fisheries was the waved albatross. Waved albatrosses constituted 65% of all seabird bycatch (n = 11) in the demersal longline fishery for hake and 50% of all seabird bycatch (n = 2) in the surface longline fishery for tuna (Table 3, S2). Black petrels (*Procellaria parkinsoni*) and blue-footed boobies (*Sula nebouxii*) were also taken in both fisheries and pink-footed shearwaters (*Puffinus creatopus*) were captured in the demersal hake fishery.

Thirty-eight seabirds were observed captured in the Peruvian surface longline fishery targeting sharks and rays. Albatrosses were the most frequent bycatch species in this fishery with 22 birds captured (Table 3, S2). The black-browed albatross was the most frequent bycatch species (n = 18) constituting 47% of total seabird bycatch and 82% of all albatrosses captured. Three other albatross species were also observed captured: Chatham (*Thalassarche eremite*), grey-headed (*Thalassarche chrysostoma*) and Buller's (*Thalassarche bulleri*). White-chinned petrels (*Procellaria aequinoctialis*) were the second most common seabird bycatch species in this fishery with 14 total captures. White-chinned petrels were also the majority of the bycatch in the Peruvian surface longline fishery targeting dolphinfish where they were 4 of the 7 seabirds observed captured (Table 3, S2).

The Peruvian surface driftnet fishery targeting sharks and rays had the highest number of observed seabird bycatch events (n = 49) and the greatest diversity of bycatch species (at least 12 species observed captured) (Table 3, S2). Guanay cormorants (*Phalacrocorax bougainvillii*) and white-chinned petrels made up the majority of total bycatch with 14 and 12 captures, respectively. One waved albatross, two black-browed albatrosses and one grey-headed albatross were also observed captured. Other seabird bycatch species included the Peruvian booby (*Sula variegata*), blue-footed booby, Inca tern (*Larosterna inca*), Humboldt penguin (*Spheniscus humboldti*), pink-footed shearwater and sooty shearwater (*Puffinus griseus*) (Table 3, S2). In addition to those seabirds captured incidentally, during 7 fishing trips another 25 waved albatrosses and 4 white-chinned petrels were intentionally captured using a baited hook and line for consumption by the crew during the trip.

3.2 Fates

Seabird mortality was highest in the Peruvian driftnet fishery with 87% of seabird bycatch recovered dead and either discarded or used for food or bait (Table 2). This compares with mortality rates of 55% in the Peruvian shark surface longline fleet, 44% in the Ecuadorian South Pacific hake demersal longline fleet, 25% in the Ecuadorian yellowfin tuna surface longline fleet and 14% in the Peruvian dolphinfish longline fleet (Table 2). Seabirds recovered alive were typically released alive while those recovered dead were usually discarded. The most frequent hooking location across all longline fisheries was in the throat while wing entanglements were the most frequent cause of capture in driftnets.

3.3 Spatial and temporal distribution

The spatial distribution of bycatch reflects fleet fishing locations (Fig. 1). Seabird bycatch in the Peruvian longline and driftnet fleet was largely clustered over the continental shelf or break, primarily in the waters offshore the ports of Salaverry and Ilo. Seabird bycatch in the Ecuadorian demersal hake fishery was concentrated in a very small area reflecting the precise fishing locations targeted by vessels in this fishery. Bycatch locations for the surface tuna fleet were more broadly distributed throughout the fishing area.

The most notable seasonal pattern to bycatch observed was for the waved albatross. Bycatch of this species in the Ecuador longline fisheries for tuna and hake was limited to the months of August to October and December (Fig. S1a,b). White-chinned petrel captures in the Peruvian fisheries were observed only in the months of June through December (Fig. S1c,d).

3.4 Bycatch rates and annual estimates

Among the four longline fleets with seabird bycatch, the Ecuador surface fishery for tuna had the highest CPUE at 0.237 ± 0.339 seabirds 1000 hooks^{-1} (range: 0 to 0.926, n = 110), followed by the Ecuadorian demersal fishery for hake at 0.112 ± 0.224 seabirds 1000 hooks^{-1} (range: 0 to 0.741, n = 417), the Peru surface fishery for sharks at 0.076 ± 0.065 seabirds 1000 hooks^{-1} (range: 0 to 0.167, n = 651) and the Peru surface fishery for dolphinfish at 0.021 ± 0.043 seabirds 1000 hooks^{-1} (range: 0 to 0.133, n = 591) (Table 2, S3). The Peruvian driftnet fishery for sharks and rays had a seabird CPUE of 0.159 ± 0.179 (range: 0 to 0.475, n = 914). In each of these fisheries approximately 10% of trips and 3% of sets fisheries had seabird bycatch, except for the Peruvian longline fishery for dolphinfish whose rates were 4.5% of trips and 1% of sets (Table 2, S3).

Available national level fishing effort data indicate that Peruvian longline fishing vessels set approximately 80 million hooks annually and Peruvian net vessels conduct approximately 80 000 fishing trips annually. National level effort data are not available for Ecuador so fishing effort estimates were developed during this study. We were able to estimate the total fleet size for the hake fishery at 80 vessels but have conservatively based our estimates on 50 active vessels. National level effort data were not available for the yellowfin tuna fishery but we were able to assess the number of vessels operating out of Santa Rosa port. We estimate that the Ecuadorian longline tuna and hake fisheries in Santa Rosa both set approximately 3 million and 8 million hooks per annum, respectively. These values are derived from the number of fishing vessels known to be actively fishing combined with data on their average number of fishing trips, sets per trip and hooks per set (Table 1, S1). Given the CPUE rates reported above and the fishing efforts listed here we estimate that the Ecuadorian hake fishery catches annually approximately 1000 seabirds, the yellowfin tuna fishery of Santa Rosa catches approximately 500 seabirds, Peruvian longline fisheries catch a combined total of approximately 5000 seabirds per annum, and Peruvian driftnet fisheries catch approximately 10 000 seabirds per annum (Table 3).

Discussion

Studies of seabird bycatch have largely been limited to large-scale, industrial fisheries with limited research available on catch rates or at-risk species associated with small-scale or coastal fisheries (Anderson et al. 2011; Bugoni et al. 2008a; Bugoni et al. 2008b; Lewison et al. 2004a; Soykan et al. 2008; Zydelis et al. 2009). Moreover studies of seabird bycatch have largely focused on the impacts of longline and trawl fisheries while information on the impacts of net fisheries has lagged (Bugoni et al. 2008a; Bugoni et al. 2008b; Zydelis et al. 2009). Here we provide onboard observer based estimates of seabird catch rates in a range of small-scale longline and driftnet fisheries in the southeastern Pacific Ocean. It has shown that seabird bycatch in these fisheries occurred regularly, often at high rates, and included numerous species of regional or global concern given their designation as critically endangered, endangered, threatened, near threatened or vulnerable. This work has also shown that it is possible to use onboard observers to rigorously monitor small-scale fisheries and their interactions with seabirds.

4.1 Seabird bycatch rates

The highest longline bycatch rate of seabirds observed in this study was for the Ecuadorian surface yellowfin tuna fishery (0.237 per 1000 hooks). This rate was about double that observed for the Ecuadorian hake fishery and three to ten times that observed for the Peruvian longline shark and dolphinfish fisheries, respectively. The catch rates observed here are of similar orders of magnitude to those observed in other longline fisheries globally, including large-scale fisheries (Anderson et al. 2011). The bycatch rate for the Ecuadorian yellowfin tuna fishery is one of the highest reported bycatch rates and is similar to that of the Brazilian southwest Atlantic pelagic longline fishery (Bugoni et al. 2008a) and the Japanese high seas southern Bluefin tuna fleet (Minami et al. 2009). Comparison of the bycatch rate for the Peruvian driftnet shark fishery to other net fisheries is made difficult by the variety CPUE metrics used (e.g. catch per net pane; per fishing day; per fishing day km; per 100 nets; per km^2 net) in other reported studies (e.g. Benjamins et al. 2008; Cardoso et al. 2011; Carretta et al. 2004; Zydelis et al. 2009) and catch estimates vary widely, but our estimated catch is broadly similar to that of the Newfoundland and Labrador gillnet fisheries (Benjamins et al. 2008) which reported an annual estimated catch of approximately 5 000 to 10 000 seabirds. Bycatch in the Peruvian driftnet fishery was also significant due to the high observed mortality rate (80%) and the fact that this was the only fishery in which a large percentage of seabird bycatch was retained for human consumption.

4.2 Species of concern

Unlike other albatross species, waved albatrosses were previously thought not to exhibit ship following behavior and therefore to be less at risk from fisheries bycatch (Anderson et al. 1997). Scavenging behavior has, however, been reported for the species and was considered a potential risk with regard to fishery interactions (Merlen 1996). Awkermen et al. (2006) reported declines in annual adult survival and evidence that fisheries bycatch and targeted take may be negatively impacting the population. These observed population declines were later re-affirmed and are indicative of a shrinking population (Anderson et al. 2008). Satellite tracking of breeding adults from the primary nesting colony in the Galapagos Islands shows that they move to coastal southern Ecuador and northern and central Peru to forage, primarily over the continental shelf (Anderson et al. 1997; Anderson et al. 2003). In the present study, the waved albatross was the seabird species captured by the greatest number of observed fishing fleets (3 of 5), was captured in both longline and gillnet fishing gears, and was the most frequent bycatch species in the Ecuadorian fisheries (62% of observed seabird bycatch). This represents the first documented bycatch of waved albatrosses by Ecuadorian fisheries and represents a previously unreported source of mortality for the species. Taken together, these sources of mortality (incidental capture and targeted take) reported here and in previous studies (Awkerman et al. 2006) may help in better understanding the reasons for the observed declines in adult survival.

While there were no observed captures of waved albatrosses by Peruvian longline vessels, these vessels did have bycatch of four other albatross species. This suggests that longline bycatch of waved albatrosses may occur, but more intense observer effort within the waved albatross distribution would be necessary. Moreover, observer effort in the Peruvian longline fleet occurred primarily out of the southern port of Ilo which is toward the limit of the known foraging range of the species (Anderson et al. 1997; Anderson et al. 2003). To a degree, however, the relatively coastal foraging distribution of species in Peru may serve to naturally segregate them from longline fishing grounds which are typically in more oceanic waters (Alfaro-Shigueto et al. 2011; Alfaro-Shigueto et al. 2010; Mangel et al. 2010).

Also observed as bycatch during the study were numerous other species of regional or global conservation concern including the Chatham, black-browed, grey-headed and Buller's albatrosses, black petrels, white-chinned petrels, Humboldt penguins and pink-footed shearwaters. While some of these species are endemic (e.g. Humboldt penguin, guanay cormorant), others have traveled across the southern Pacific Ocean to forage (e.g. black petrel, Chatham albatross) or are migrating along the Pacific coast of the Americas between nesting and foraging habitats (e.g. pink-footed shearwater) (Freeman et al. 2010; Guicking et al. 2001; Pitman and Ballance 1992; Spear et al. 2003). For some of these species, like the Humboldt penguin (Majluf et al. 2002; Simeone et al. 1999), previously studies have reported fisheries bycatch, while for others, like the pink-footed shearwater and black petrel, given their distributions and foraging grounds, there was concern that bycatch could occur, but evidence was lacking (CEC 2005; Francis and Bell 2010; Guicking et al. 2001). Our research re-affirms that bycatch of these seabird species should be taken into account when setting future research and conservation priorities (CEC 2005; Francis and Bell 2010).

4.4 Regional impact

Small-scale fisheries in Ecuador and Peru are important sources of food and employment in many coastal communities (Alfaro-Shigueto et al. 2010; Alvarez 2003; Pinoargote 2008) and these fisheries have experienced substantial growth and diversification within the past decades (Alfaro-Shigueto et al. 2010; CPPS 1986; Estrella and Swartzman 2010; Pinoargote 2008). In this study we have identified five fisheries with seabird bycatch and we estimate that the annual bycatch in these fisheries ranges from approximately 500, as in the case of Ecuadorian longline yellowfin tuna fisheries in Santa Rosa port to about 5 000 and 10 000 for the Peruvian longline and driftnet shark fisheries, respectively. The elevated catch rates observed in the Ecuadorian longline fisheries are of particular concern because the primary bycatch species is the critically endangered waved albatross. Moreover, these estimates apply to only select fishing fleets while Ecuador has approximately 150 total ports and landing sites and an artisanal fleet of approximately 15 000 vessels (Arriaga and Martinez 2003a; Pinoargote 2008). The region of Esmereldas in southern Ecuador alone, where fisheries overlap with waved albatrosses is likely highest (Anderson et al. 1997; Anderson et al. 2003), has an estimated 6 000 small-scale vessels in operation (Pinoargote 2008). Peru's small scale fisheries are similarly large. Longline fisheries are the fastest growing sector of Peru's small-scale fisheries (Alfaro-Shigueto et al. 2010; Estrella and Swartzman 2010) but net fisheries remain the largest component of Peru's small-scale fleet and this sector also continues to expand (Alfaro-Shigueto et al. 2010). Our work presents a snapshot of these fisheries sectors for several monitored ports and there will certainly be variations in bycatch rates and species composition between gear types and as one moves along the coast. However, given the observed bycatch rates, annual seabird bycatch estimates and the size and diversity of the small-scale fishing fleets of Ecuador and Peru, there is clearly an urgent need for a more thorough assessment of small-scale fishing effort at both the national and regional level and an assessment of the potential impacts, spatio-temporal patterns and species composition of marine vertebrate bycatch.

4.5 Mitigation opportunities

Considerable work has been done trialing seabird bycatch mitigation measures (Bull 2007; Gilman et al. 2005; Melvin et al. 1999). These methods, which include varying fishing practices (e.g. time of set, offal management) or the use of mitigation measure such as tori lines, increased line weighting, side-setting and line-shooters, have focused largely on large-scale longline and trawl fisheries (Bull 2007; Gilman et al. 2005; Gilman et al. 2008b; Sullivan et al. 2006a). Efforts to mitigate seabird bycatch in net fisheries has proven more challenging to address but a number of measures have been proposed including enhancing net visibility, use of acoustic alarms and increasing net depth (Bull 2007; Melvin et al. 1999; Zydelis et al. 2009). Implementation of mitigation measures in small-scale fisheries faces considerable challenges because these fisheries are often characterized by being economically marginalized, highly dispersed, open-access, and prone to poor management or enforcement (Chuenpagdee et al. 2006; Jacquet and Pauly 2008; Salas et al. 2007). The small size of these fishing vessels can also make some mitigation measures impractical (e.g. limited storage space or vessel height). Under these conditions, which are broadly applicable to the fisheries we monitored in Ecuador and Peru, it is difficult to mandate the purchase or ensure the proper use of mitigation measures. Mitigation in the Peruvian driftnet fishery could prove particularly challenging as it can be considered an entry-level fishery (Alfaro-Shigueto et al. 2010), mortality rates were high and a large proportion of bycatch we observed was eaten. Ideally, mitigation measures should be inexpensive, constitute a permanent fix or change to the fishing gear and not reduce target catch rates. Work currently underway trialing gear and line weighting changes in the Ecuadorian hake fishery and introducing weighted swivels to Peruvian longline fisheries serve as examples of relatively inexpensive means to potentially reduce seabird bycatch.

4.6 Future directions

The small-scale fisheries monitored in this study varied in terms of fishing methods, spatial and temporal distributions, and target species. Moreover, observer effort in these fisheries varied, with some monitored for significantly longer periods than others. Nevertheless, we have shown the effectiveness of onboard observer programs in assessing seabird bycatch in small-scale fisheries. Future work in these fisheries and with the seabirds species with which they interact can benefit from the baseline seabird bycatch rates developed here. Given the catch rates we report and the sizes of the fisheries involved, it is clear that there is an urgent need, both in the southeastern Pacific region and globally, for further efforts to better understanding the impacts of small-scale fisheries on seabirds (Alfaro-Shigueto et al. 2010). Within the waters of Ecuador and Peru, a more thorough accounting of these fisheries will be particularly important toward promoting the recovery of the critically endangered waved albatross (Anderson et al. 2008; Awkerman et al. 2006). Additional work identifying, trialing and implementing low cost mitigation solutions is currently underway in Ecuador and Peru but more work is needed. The creation of seabird-national plans of action (NPOA-Seabirds) could also help set research and conservation priorities and is strongly encouraged as is the continued active participation in international agreements promoting seabird conservation like the Agreement on the Conservation of Albatrosses and Petrels (ACAP).

Acknowledgements

We warmly thank all the participating boat owners, fishing captains and crew for their generous collaboration, without which this project would not have been possible. We also thank the Ecuadorian Ministry of the Environment and the Provincial government of Santa Elena, Ecuador for their support. We thank onboard observers Eduardo Merejildo, George Marin, Robert Medina, Leonidas Luzardo, Geovanny Sandoval, Fernando Palacios, and Roque Cedeño for their excellent work. We also thank Pro Delphinus staff for assistance with data collection and management. Nigel Brothers also provided invaluable assistance and expertise in project design and implementation and we thank him for his assistance. Ecuador project funding was provided by American Bird Conservancy and the David and Lucille Packard Foundation. Peru project funding was provided by Agreement on the Conservation of Albatrosses and Petrels, American Bird Conservancy, Conservation Leadership Programme, David and Lucille Packard Foundation, Darwin Initiative, International Association of Antarctica Tour Operators, NOAA-National Marine Fisheries Service, and Peregrine Fund. We also acknowledge the generous support for this work by the governments of Peru and Ecuador through ACAP. JCM and JAS are Overseas Research and Student Award Scheme (ORSAS) and University of Exeter scholarship awardees, respectively. MJW is funded by the Peninsula Research Institute for Marine Renewable Energy (PRIMaRE). DH is supported by the Natural Environment Research Council and the European Union. BJG receives funding from the Darwin Initiative, the Natural Environment Research Council and the European Union.

Table 1. Summary of fisheries and observer effort by country. Port abbreviations are Santa Rosa (SR), Anconcito (An), Ilo and Salaverry (Sal). Fleet sizes are for these specific ports.

Country	Fishery	Target species	Port	Fleet Size	Observed trips	Observed sets	Observed Hooks / net	Fishing days	Monitoring Period	Fishing season	Seabird bycatch
Ecuador	Surface longline	Yellowfin tuna	SR	150	47	110	20,807	97	05/09 – 11/10	Jul - Dec	Yes
	Surface longline	Dolphinfish	SR	200	27	64	25,880	55	11/08 – 04/10	Dec – Mar	No
	Midwater longline	Escolar	SR	100	56	111	48,082	111	02/09 – 10/10	Jun - Dec	No
	Demersal longline	South Pacific hake	SR	25	127	417	165,818	128	09/09 – 04/11	Year-round	Yes
	Demersal longline	Pacific bearded brotula	An	5	17	33	69,682	17	08/10 – 11/10	Year-round	No
	Driftnet	Skipjack tuna Striped bonito Dolphinfish	SR	850	80	165	238 km	160	11/08 – 04/11	Year-round	No
Peru	Surface longline	Blue shark Short-fin mako	Ilo/Sal	100	87	651	536,158	647	05/05 – 05/11	Mar - Nov	Yes
	Surface longline	Dolphinfish	Ilo/Sal	250	89	591	601,840	544	05/05 – 05/11	Dec - Feb	Yes
	Driftnet	Blue shark Short-fin mako Thresher shark Smooth hammerhead Eagle ray	Sal	60	133	914	1738 km	799	05/05 – 05/11	Year-round	Yes

Table 2. Summary by country and fishery of the mortality rates, capture times, locations of hooking or entanglement and final uses of all seabirds captured during the study. Fisheries are listed by gear type (longline or net) and their main target species. Observed seabird catch per unit effort (CPUE; mean and 95% CI) by country, fleet and target species. Catch per unit effort is presented per 1000 hooks for longline fisheries and per km^{2}h for the driftnet fishery. Hook location categories are beak (Be), throat (Th), chest (Ch), wing (Wi) and other (Oth). Bycatch use categories are released alive (Rel), discarded dead (Dis), eaten onboard (Eat) and other (Oth). Seabird type codes are black-browed albatross (BBAL), waved albatross (WVAL) and white-chinned petrel (WCPE).

			Catch	CPUE		% with bycatch		Hook location (%)					Use (%)			
Country	Fishery	Seabird type	quantity	1000 h / km^{2}h	Range	Trips	Sets	Be	Th	Ch	Wi	Oth	Rel	Dis	Eat	Oth
Ecuador	Longline:	Total	17	0.112 ± 0.224	(0.0 – 0.741)	11.0	3.4	0	100	0	0	0	56	44	0	0
	Hake	WVAL	11	0.036 ± 0.070	(0.0 – 0.206)			0	100	0	0	0	60	40	0	0
		Not WVAL	6	0.076 ± 0.213	(0.0 – 0.741)			0	100	0	0	0	50	50	0	0
	Longline:	Total	4	0.237 ± 0.339	(0.0 – 0.926)	8.5	3.6	25	50	0	25	0	75	25	0	0
	Tuna	WVAL	2	0.139 ± 0.312	(0.0 – 0.926)			0	50	0	50	0	100	0	0	0
Peru	Longline:	Total	38	0.076 ± 0.065	(0.0 – 0.167)	12.6	3.7	42	50	3	0	6	45	50	0	5
	Shark &	Albatross	22	0.044 ± 0.042	(0.0 – 0.102)			36	50	5	0	9	50	50	0	0
	ray	BBAL	18	0.037 ± 0.033	(0.0 – 0.097)			44	44	6	0	6	56	44	0	0
		WCPE	14	0.028 ± 0.023	(0.0 – 0.057)			46	54	0	0	0	36	50	0	14
	Longline:	Total	7	0.021 ± 0.043	(0.0 – 0.133)	4.5	0.85	25	75	0	0	0	86	14	0	0
	Dolphinfish	WCPE	4	0.019 ± 0.044	(0.0 – 0.133)			25	75	0	0	0	75	25	0	0
	Driftnet:	Total	49	0.159 ± 0.179	(0.0 – 0.475)	10.5	2.5	11	11	11	47	21	13	48	35	4
	Shark &	Albatross	4	0.015 ± 0.028	(0.0 – 0.084)			50	50	0	0	0	67	0	33	0
	ray	Not Albatross	45	0.144 ± 0.177	(0.0 – 0.475)			6	6	12	53	24	9	48	33	4
		BBAL	2	0.007 ± 0.017	(0.0 – 0.046)			100	0	0	0	0	50	0	50	0
		WCPE	12	0.032 ± 0.049	(0.0 – 0.131)			0	13	13	75	0	8	92	0	0

Table 3. Annual seabird catch estimates for observed fisheries and noting particular species of concern given their frequency as bycatch or their regional or global conservation status. Annual effort and observed catch rates (CPUE) are derived from this study or from published reports of fleet-wide fishing effort. Sources of national level fishing effort data for Peruvian fisheries are: Alfaro-Shigueto et al. 2010; Escudero 1997; Estrella 2007; Estrella et al. 1999; Estrella et al. 2000. Effort data for Ecuadorian fisheries were obtained during this study. Hake fishery data are national level while tuna fishery data are for the port of Santa Rosa only. Values in square brackets are extrapolations to allow for calculation of catch estimates based upon the available metrics of fishing effort.

Country	Fishery	National level Fishing effort	Observer catch per unit effort	Seabird Catch estimate	Species Of concern
Ecuador	Longline: Hake	50 vessels [6000 trips] [8 000 000 hooks]	0.112 per 1000 hooks	1000	Waved albatross Black petrel Pink-footed shearwater
(Santa Rosa only)	Longline: Tuna	150 vessels [6000 trips] [3 000 000 hooks]	0.237 per 1000 hooks	500	Waved albatross Black petrel
Peru	Longline: Shark & dolphinfish	80 000 000 hooks	Shark: 0.076 per 1000 hooks Dolphinfish: 0.021 per 1000 hooks [Seasonally weighted: 0.062 per 1000 hooks]	5 000	Black-browed albatross Chatham albatross White-chinned petrel
	Driftnet: Shark & ray	80 000 total net trips [24 000 driftnet trips]	0.159 per km^2h [0.31 per trip]	10 000	Waved albatross Pink-footed shearwater Humboldt penguin Guanay cormorant White-chinned petrel

Figure 1. Fisheries sampled and locations and relative quantities of seabird bycatch. Shaded areas are minimum convex polygons (MCP) of locations of observed fishing sets for the monitored fisheries. Ecuador fisheries were confined to the MCP off the port of Santa Rosa, Ecuador. The port of Anconcito is very near to Santa Rosa thus not visible. Observed sets of the Peru driftnet shark fishery extended approximately to the 200m bathymetric contour in the waters fronting Salaverry port. Locations of seabird bycatch are represented by the black circles. Seabird bycatch species composition for each fishery is indicated in a pie chart. Fishery abbreviations indicate gear type (longline [LL], or net) and main target species.

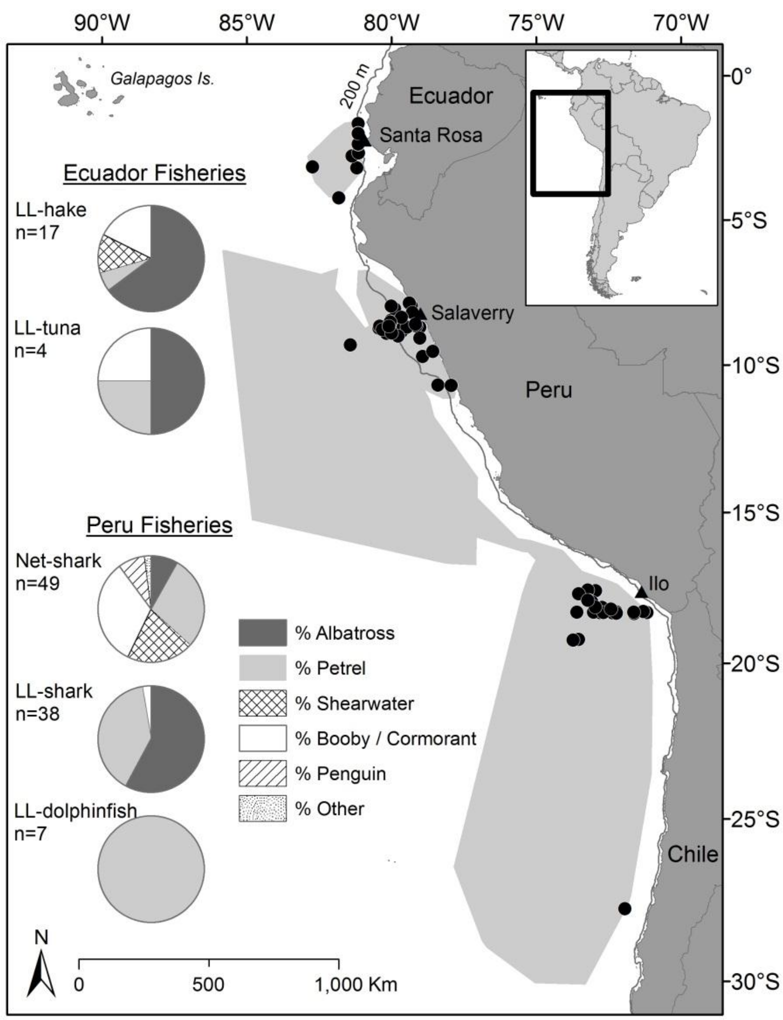


Table S1. Fisheries characteristics and observed effort summary for each of the fisheries monitored during the study, 2005 to 2011.

	Ecuador						Peru		
Gear type	Longline	Longline	Longline	Longline	Longline	Driftnet	Longline	Longline	Driftnet
Target species	Yellowfin tuna	Dolphinfish	Escolar	South Pacific Hake	Pacific bearded Brotula	Skipjack tuna Striped bonito Dolphinfish	Blue shark Mako shark	dolphinfish	Blue shark Mako shark Hammerhead Thresher shark Eagle ray
Seabird bycatch?	Yes	No	No	Yes	No	No	Yes	Yes	Yes
Monitoring period	05/09 to 11/10	11/08 to 04/10	02/09 to 10/10	09/09 to 04/11	08/10 to 11/10	11/08 to 04/11	05/05 to 05/11	05/05 to 05/11	05/05 to 05/11
Fishing season	Jul to Dec	Dec to Mar	Jun to Dec	Year-round	Year-round	Year-round	Mar to Nov	Dec to Feb	Year-round
Trips	47	27	56	127	17	80	87	89	133
Sets	110	64	111	417	33	165	651	591	914
Fishing days	97	55	111	128	17	160	647	544	799
Set depth	Surface	Surface	Midwater	Demersal	Demersal	Surface	Surface	Surface	Surface
Sets per trip	2.3	2.4	2.0	3.3	1.9	2.1	7.5	6.6	6.9
Hooks/net observed	20 807	25 880	48 082	165 818	69 682	238 km	536 158	601 840	1738 km
Hook type	7/0, 8/0	J4, J5	J4, J5	J9	J9, J10	n/a	J0, J1, J2	J4, J5	n/a
Set time	Morning	Throughout day	Late evening	Throughout day	Evening	Late afternoon	Morning	Throughout day	Late afternoon
Set duration (h)	7.4 ± 3.8 (0.8 – 18.8)	5.9 ± 2.5 (2.4 - 11.9)	6.8 ± 1.2 (4.0 – 10.4)	0.7 ± 0.2 (0.1 – 2.6)	2.0 ± 0.9 (0.6 – 4.6)	11.2 ± 2.1 (2.6 - 15.5)	12.9 ± 3.9 (1.0 – 35.4)	7.4 ± 2.6 (2.4 - 20.7)	13.4 ± 3.6 (1.3 – 34.8)
Bait	Squid Mackerel	Squid herring	Squid	Squid harvestfish	Sardine Herring	n/a	Squid Mackerel mullet	Squid Mackerel mullet	Small cetacean
Hooks per set	189 ± 68 (80 – 460)	415 ± 105 (120 – 650)	441 ± 68 (300 – 560)	397 ± 76 (280 – 675)	2111 ± 1282 (420 – 4000)	n/a	824 ± 353 (260 - 2000)	1018 ± 485 (200 - 2000)	n/a
Net length (km)	n/a	n/a	n/a	n/a	n/a	1.5 ± 0.2 (1.1 - 1.8)	n/a	n/a	1.9 ± 0.5 (0.7 – 5.3)
Mesh size (in)	n/a	n/a	n/a	n/a	n/a	5	n/a	n/a	7.5, 8, 10

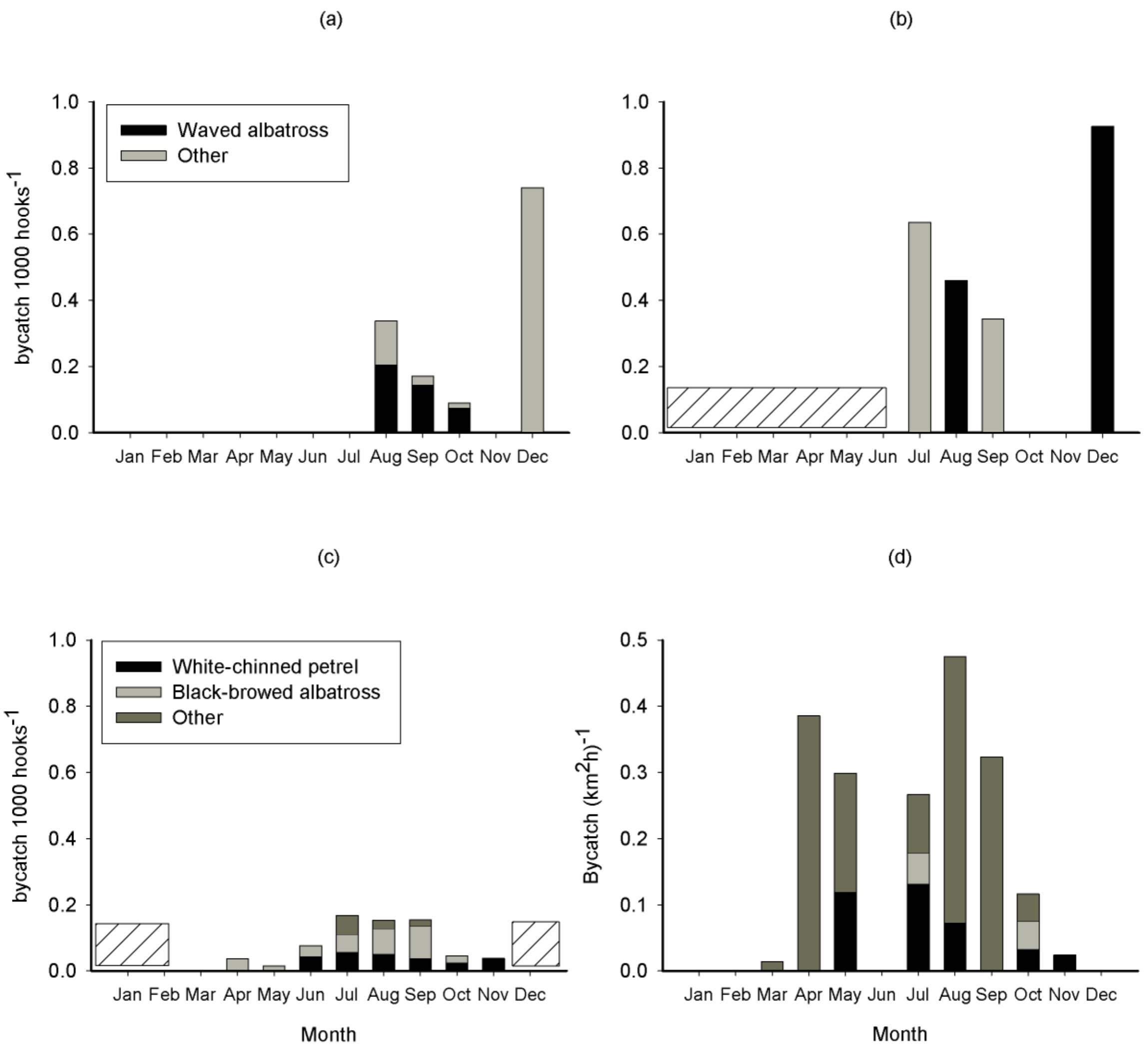
Table S2. Summary of seabird bycatch by country and fishery. Fishery abbreviation “LL” refers to longline fisheries. Capture species IUCN conservation status is listed in parenthesis: CR = critically endangered, EN = endangered, NT = near threatened, VU = vulnerable, LC = least concern.

		Ecuador		Peru		
Common name	Latin name	LL-demersal for hake	LL-surface for tuna	Driftnet for shark & ray	LL-surface shark & ray	LL-surface for dolphinfish
Waved albatross (CR)	*Phoebastria irrorata*	11	2	1	-	-
Black-browed albatross (EN)	*Thalassarche melanophris*	-	-	2	18	-
Chatham albatross (VU)	*Thalassarche eremite*	-	-	-	1	-
Grey-headed albatross (VU)	*Thalassarche chrysostoma*	-	-	1	1	-
Buller’s albatross (NT)	*Thalassarche bulleri*	-	-	-	2	-
White-chinned petrel (VU)	*Procellaria aequinoctialis*	-	-	12	14	4
Black petrel (VU)	*Procellaria parkinsoni*	1	1	-	-	-
Storm petrel	*Oceanodroma* spp.	-	-	-	1	3
Unidentified petrel	-	-	-	2	-	-
Peruvian booby (LC)	*Sula variegate*	-	-	1	-	-
Blue-footed booby (LC)	*Sula nebouxii*	3	1	1	-	-
Guanay cormorant (NT)	*Phalacrocorax bougainvillii*	-	-	14	-	-
Southern skua (LC)	*Catharacta antarctica*	-	-	-	1	-
Inca tern (NT)	*Larosterna inca*	-	-	1	-	-
Humboldt penguin (VU)	*Spheniscus humboldti*	-	-	4	-	-
Pink-footed shearwater (VU)	*Puffinus creatopus*	2	-	4	-	-
Sooty shearwater (NT)	*Puffinus griseus*	-	-	6	-	-

Table S3. Seabird catch per unit effort (CPUE) comparison table. Bycatch categories include total catch for each fishery observed and for particular species or species groups. Species codes are: waved albatross (WVAL), black-browed albatross (BBAL), white-chinned petrel (WCPE). Fisheries are described by their gear type (net or longline) and by their main target species. CPUE estimates are averages of pooled monthly seabird bycatch during the study period, 2005 to 2011.

			Per set		Per 1000 hooks		Per (1000 hooks * h)		Per Km^{2}h	
Country	Fishery	Bycatch category	Mean ± SD	Range	Mean ± SD	Range	Mean ± SD	Range	MEAN	Range
Ecuador	Longline:	Total bycatch	0.038 ± 0.070	(0.0 - 0.222)	0.112 ± 0.224	(0.0 - 0.741)	0.159 ± 0.310	(0.0 - 1.010)	n/a	n/a
	Hake	WVAL	0.014 ± 0.026	(0.0 - 0.075)	0.036 ± 0.070	(0.0 - 0.206)	0.054 ± 0.108	(0.0 - 0.322)	n/a	n/a
		Not WVAL	0.025 ± 0.064	(0.0 - 0.222)	0.076 ± 0.213	(0.0 - 0.741)	0.105 ± 0.290	(0.0 - 1.010)	n/a	n/a
	Longline:	Total bycatch	0.036 ± 0.047	(0.0 - 0.111)	0.237 ± 0.339	(0.0 - 0.926)	0.036 ± 0.075	(0.0 - 0.243)	n/a	n/a
	Tuna	WVAL	0.019 ± 0.040	(0.0 - 0.111)	0.139 ± 0.312	(0.0 - 0.926)	0.028 ± 0.076	(0.0 - 0.243)	n/a	n/a
Peru	Longline:	Total bycatch	0.049 ± 0.041	(0.0 - 0.114)	0.076 ± 0.065	(0.0 - 0.167)	0.004 ± 0.003	(0.0 - 0.009)	n/a	n/a
	Shark & ray	All albatrosses	0.031 ± 0.030	(0.0 - 0.071)	0.044 ± 0.042	(0.0 - 0.102)	0.003 ± 0.002	(0.0 - 0.006)	n/a	n/a
		BBAL	0.024 ± 0.021	(0.0 - 0.068)	0.037 ± 0.033	(0.0 - 0.097)	0.002 ± 0.002	(0.0 - 0.005)	n/a	n/a
		WCPE	0.019 ± 0.016	(0.0 - 0.038)	0.028 ± 0.023	(0.0 - 0.057)	0.002 ± 0.001	(0.0 - 0.004)	n/a	n/a
	Longline:	Total bycatch	0.025 ± 0.054	(0.0 - 0.167)	0.021 ± 0.043	(0.0 - 0.133)	0.003 ± 0.005	(0.0 - 0.016)	n/a	n/a
	Dolphinfish	WCPE	0.023 ± 0.054	(0.0 - 0.167)	0.019 ± 0.044	(0.0 - 0.133)	0.002 ± 0.005	(0.0 - 0.016)	n/a	n/a
	Net:	Total bycatch	0.061 ± 0.069	(0.0 - 0.176)	n/a	n/a	n/a	n/a	0.159 ± 0.179	(0.0 - 0.475)
	Shark & ray	All albatrosses	0.004 ± 0.009	(0.0 - 0.027)	n/a	n/a	n/a	n/a	0.015 ± 0.028	(0.0 - 0.084)
		Not albatrosses	0.057 ± 0.068	(0.0 - 0.176)	n/a	n/a	n/a	n/a	0.144 ± 0.177	(0.0 - 0.475)
		BBAL	0.002 ± 0.006	(0.0 - 0.016)	n/a	n/a	n/a	n/a	0.007 ± 0.017	(0.0 - 0.046)
		WCPE	0.013 ± 0.020	(0.0 - 0.056)	n/a	n/a	n/a	n/a	0.032 ± 0.049	(0.0 - 0.131)

Figure S1. Mean pooled monthly bycatch of select seabird species in the (a) Ecuadorian demersal longline fishery for hake, (b) Ecuadorian surface longline fishery for yellowfin tuna, (c) Peruvian surface longline fishery for sharks and rays, (d) Peruvian surface driftnet fishery for sharks and rays. Seabird bycatch per unit effort is presented as catch per 1000 hooks for longline fisheries and catch per km^2h for the driftnet fishery. Seasonal fisheries (b) and (c) have their out-of-season periods indicated by the cross-hatched polygons.



Discussion

Project summary

In this thesis I have presented a series of works assessing the interactions of marine vertebrates with small-scale fisheries and fishing communities along the Pacific coast of South America. The majority of the work was conducted in Peru but several chapters were possible only through coordination and collaboration with fellow researchers in Ecuador and Chile and present results for projects spanning large portions of the southeastern Pacific Ocean region. The chapters I present are on diverse themes and entail research with marine mammals, sea turtles and seabirds and with small-scale longline and gillnet fisheries. This diversity of topics is representative of the range of research and conservation issues that presently face marine conservation and efforts to better understand the dynamics and impact of small-scale fisheries, and it is representative of the broad skill set researchers and conservationists must have in order to effect change.

In chapter 1, “Post-capture movements of loggerhead turtles in the southeastern Pacific Ocean assessed by satellite tracking” I showed that loggerhead turtles were resident over extended time periods and spatial areas in pelagic waters off Peru and Chile, waters used heavily by Peruvian small-scale longline fisheries thus putting these turtles at repeated risk of interaction. In chapter 2, “Latitudinal variation in diet and patterns of human interactions in the marine otter”, I reported on the use by marine otters of coastal fishing communities as den and foraging sites. While beneficial from the animal’s standpoint as excellent den sites and sources of easily accessible food, this behavior also puts the animals at risk due to interactions with rats and feral cats and dogs as well as at risk of entanglement with discarded fishing gear. However, if these coastal communities were managed properly marine otters could live safely alongside humans and these locations could serve as stepping stones along the coast. I also report on a latitudinal variation in diet that shows marine otters in Peru foraging more on fish species than marine otters to the south. This also has implications regarding population viability as it could put the species in more direct competition with fishermen for their prey. Chapter 3, “Small cetacean captures in Peruvian artisanal fisheries: High despite protective legislation” and Chapter 4, “Using pingers to reduce small cetacean bycatch in the small-scale driftnet fishery in Peru”, show the progression of work from problem characterization to trials of a potential mitigation measure. In chapter 3 I report on the results of using onboard and shore-based observers to assess small cetacean bycatch in

Peruvian longline and gillnet fisheries. The results made clear that small cetacean bycatch in Peru continued at a very high rate in spite of the existence of legislation banning the take of small cetaceans. Given this circumstance of continued take and minimal enforcement, in Chapter 4 I report on a trial to reduce small cetacean bycatch in the Peruvian driftnet fishery using acoustic alarms (pingers). Pingers were effective and reduced the total catch of small cetaceans by 37% while not reducing target catch rates. And it is clear that there is interest among fishermen to use pingers. However it is also clear that considerable challenges remain toward reducing small cetacean interactions as they are still a common source of bait (obtained from bycatch and harpooning) and Burmeister's porpoises remain an accepted food source. In Chapter 5, "Onboard observer data suggest that small-scale fisheries are a major potential threat to seabirds in the southeastern Pacific" I report on the results of onboard observer work in nine small-scale fisheries, seven longline fleets and two driftnet fleets. Five of the nine fleets assessed had bycatch of seabirds and this bycatch included multiple threatened and endangered species, including the critically endangered waved albatross. As one of the few studies of seabird bycatch in small-scale fisheries (and given the massive size of these fleets) the results of this work make clear the urgent need for additional work to assess seabird interactions with these fleets (and of marine vertebrates generally) in order to have a fuller understanding of the threats these species face and the potential mitigation measures that can be employed to limit or eliminate these negative interactions.

Where do we go from here?

A number of reports and publications have addressed the status of small-scale or artisanal fisheries globally (e.g. Béné 2006; Berkes et al. 2001; Chuenpagdee et al. 2006; Jacquet and Pauly 2008; McGoodwin 2001) and within the southeastern Pacific region (e.g. Alfaro-Shigueto et al. 2010; Alvarez 2003; CPPS 1986; OECD 2009). These reports make clear the size and importance of these fisheries, as sources of both food and employment. There is now also growing evidence of interactions of these fisheries with marine vertebrates, including marine mammals (e.g. Felix and Samaniego 1994; Van Waerebeek et al. 1997a), sea turtles (e.g. Alfaro-Shigueto et al. 2011; Peckham et al. 2007), seabirds (e.g. Bugoni et al. 2008a; Majluf et al. 2002; Moreno et al. 2006), and sharks (Gilman et al. 2008a) and calls for more assessment, more coordination of work and greater efforts to develop a regional or global perspective of the impacts of these fisheries on marine vertebrates (Lewison et al. 2004a; Moore et al. 2010; Soykan et al. 2008).

It is also clear that small-scale fisheries pose unique challenges to managers that set them apart from commercial or industrial fisheries. Small-scale fisheries are often open-access, widely dispersed and based at remote ports or landing sites, and subject to poor management or enforcement measures (Berkes et al. 2001; Dutton and Squires 2008; Jacquet and Pauly 2008; McGoodwin 2001). Historically this has meant that there was little research or monitoring of these fisheries. But as the reports cited above make clear, this is changing. As data on these fisheries and their bycatch becomes available it will allow us to better understand the full range of impacts to a species or population and permit the development of more accurate assessments of the potential for population management and recovery.

However management of these fisheries remains challenging as does enforcement of conservation measures. In many respects we are trying to find answers to the question, “How do we make progress in an enforcement free environment?” In this context, and as I have noted throughout this thesis, to effect change in small-scale fisheries it is essential to work alongside fishermen and within their communities and to seek mitigation measures that are inexpensive and which do not effect target catch.

Some inexpensive measures are available, such as through the introduction of weighted swivels to longline gear in Peru. Use of weighted swivels can increase the sink rate of branchlines thus reducing their availability to seabirds (Robertson et al. 2006). And these swivels represent a permanent gear change that cannot be easily removed and which also may improve fishing efficiency and reduce the workload on the crew who previously used separate swivels and weights. There is, of course, no guarantee that inexpensive, effective measures such as these can always be identified. Identification of mitigation measures for net fisheries, for example, has proven particularly challenging (Bull 2007; Gilman et al. 2010; Melvin et al. 1999; Wang et al. 2010) and these fisheries also have bycatch of multiple marine vertebrates including sea turtles (Alfaro-Shigueto et al. 2011; Gilman et al. 2010; Peckham et al. 2007), seabirds (Majluf et al. 2002; Simeone et al. 1999), marine mammals (Dawson and Slooten 2005; Jefferson and Curry 1994; Read et al. 2006) and manta rays (Alfaro-Shigueto et al. 2010).

The introduction of mitigation measures to a fishery need not be framed as punitive measures. Rather, they can be seen as attempts to make these fisheries more sustainable and this message can be used with fishermen in promoting the implementation of these measures. Global concern for species like marine mammals, seabirds and sea turtles has been made clear through the passage of numerous international conventions and agreements such as the Inter-American Sea Turtle Convention (IAC) and the Agreement on the Conservation of Albatrosses and Petrels (ACAP). By showing voluntary progress in complying with these broadly supported initiatives, small-scale fisheries can show how they are working proactively to prevent marine vertebrate bycatch and thereby also working to secure the long-term sustainability of their fisheries. There is also the potential for small-scale fisheries to work with international fishery certification bodies like the Marine Stewardship Council. There remain daunting challenges to the certification of small-scale fisheries (e.g. open access, lack of enforcement mechanisms), but if these issues can be overcome they would provide additional incentives to these fisheries to mitigate bycatch and fish more sustainably.

The multi-taxa nature of bycatch in small-scale fisheries is an additional challenge and must also be taken into account when monitoring bycatch and assessing mitigation solutions. Will a measure that reduces sea turtle bycatch increase the bycatch of seabirds? In Peru, for example, longline fisheries were reintroduced in the late 1980s in part as a means to reduce the impact of driftnet fisheries on small cetaceans. But longline fisheries are also prone to seabird and sea turtle bycatch and those same fisheries often use small cetaceans for bait. Clearly these inter-relationships between fisheries and marine fauna are complex and may take many years to elucidate. We need to be aware of this potential and take a multi-taxa view when assessing a fishery and its potential impacts on marine fauna.

Rapid assessments (Moore et al. 2010) are one tool that can help in the early detection of potential bycatch problems. In most cases, multiple, simultaneous lines of research will be necessary to fully understand a fishery. These include surveys and interviews with fishermen, the use of observers (both onboard and shore-based) and access to government economic, population and fishery statistics. By working along multiple lines of research we can often gain a fairly rigorous understanding of the at-risk species and rates of bycatch within a fishery or region. This research can also be very effective at identifying data gaps and in setting priorities for future research and conservation initiatives.

In most cases, a reduction in bycatch may be feasible, but elimination is highly unlikely. These reductions should be treated as successes and foundations upon which more progress can be built. In Peru, for example, I report in Chapter 3 that approximately 15,000 to 20,000 small cetaceans may still be taken annually as bycatch in the driftnet fishery. If we were able to reduce that take by several thousands of animals per year through the use of pingers, even in a context where some animals are still taken for use as bait or food, this would be progress. This would be progress upon which we could build additional work expanding pinger use, searching for bait alternatives and educating coastal communities and fishermen about the threats faced by these small cetacean populations.

We also need to remain vigilant to changes within these fisheries. Small-scale fisheries are extremely dynamic and capable of rapid changes to fishing methods and target species. In Peru, for example, we continue to monitor the development and mechanization of the small-scale longline fleet. As these vessels mechanize (through additions of pulleys, line pullers and mainline drums for example) they may change how they interact with the marine environment. They may double the number of hooks they deploy on a given set, or change set depth, or the setting location on the vessel, or change the bait or hook type. Any of these alterations could impact seabirds or sea turtles for example, making them more or less likely to interact with the gear. It is necessary to monitor these changes and work to anticipate impacts to marine vertebrates and prevent problems before they become engrained practices in the fishery.

This work also takes place in the context of constant, dramatic global change including climate change and human population growth. These trends place massive pressures on coastal areas as human populations surge. It also creates increasing demands for marine resources as a food source. The impacts of climate change on marine vertebrates remain muddled but one could expect impacts to marine vertebrate populations through changes to habitats and food availability and the resilience of threatened populations to respond to these changes (Bograd et al. 2011; Brander 2010; Fuentes et al. 2011; Weimerskirch et al. 2012). Research and conservation efforts with small-scale fisheries must work to hold the line against these trends. Also, as many of the developing nations with small-scale fleets continue to develop economically they may be become better able to manage their fisheries and build upon the progress made and lessons learned in the United States, European Union and elsewhere.

Education must also play a central role in any effort to effect change in small-scale fisheries. The marginalization of these coastal communities has meant that they often lack basic resources, including educational resources. Through regular, repeated contact, talks, workshops, training session, etc. with fishers, their partners and children (future fishermen) great progress can be made in raising awareness of the marine environment, marine conservation and the reasons behind conservation measures for marine vertebrates.

Marine research and conservation practitioners continue moving forward with their fisheries, nations, regions and unique circumstances. As one of these practitioners and one looking to learn from past experiences toward improving future results, I have attempted to highlight some of those lessons learned and broad patterns apparent from the chapters in this thesis. There remains much to be learned about threatened marine vertebrate species and the fisheries with which they interact. This continued work should seek to promote small-scale fishery sustainability and the continued survival of these threatened populations of marine vertebrates.

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