TRACKING GLOBAL FISHERIES FROM SPACE: PATTERNS, PROBLEMS, AND PROTECTED AREAS

by

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***“If there is magic on this planet, it is contained in water*.”**

Loren Eisele

This thesis is dedicated to all the teachers in my life,

past, present, and future,

who help me discover the wonders around me.

First and foremost, my parents and my family.

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Abstract

Fishing is one of the largest and most widespread ocean uses affecting marine ecosystems and biodiversity. From small-scale coastal vessels to industrialized high-seas fleets, the footprint of modern fisheries extends over much of the global ocean. Nonetheless, we have only limited understanding when, where, and how those fisheries are occurring, especially in remote areas far from shore. This poses a problem for fisheries management efforts and marine conservation measures such as marine protected areas (MPAs), which are being established to meet global conservation targets. Fishing is an important factor influencing the effectiveness of MPAs. It is therefore of vital importance to map and analyze the global footprint of fisheries and better understand its influences on the efficacy of marine conservation measures and fisheries management. I apply a novel satellite-based monitoring tool, the Automatic Identification System (AIS), to analyze behavior and movement patterns of fishing vessels globally in the context of marine conservation. For this I developed methods to automatically analyze fishing effort from AIS data and applied these to analyze patterns of fishing vessel behavior around the globe. These new tools allowed me to describe the global distribution of fisheries at fine spatial and temporal resolution. In some cases, fishing effort accumulated close to the boundaries of MPAs, an indicator of spillover of fish benefiting fishing fleets nearby. Near the Galápagos Marine Reserve, fishing effort within 20 km from the reserve boundary was four times higher than in the surrounding area, and tuna catches were higher and more stable near the reserve boundary as well. Patterns of fishing effort around 12 other large MPAs were shaped predominantly by their proximity to Exclusive Economic Zone and MPA boundaries, showing the major effects of maritime zoning regulations on fishing effort. Furthermore, fishing was increased around older MPAs and those in developing countries. Linking fishing vessel behavior to seafood supply chains, I also documented global patterns and hot spots of transshipment of catch to cargo vessels. Using AIS data I found transshipment particularly important in high seas fisheries, such as tuna longlining, raising concerns about mixing of legal and illegal catches in some of the world's most widespread and valuable fisheries. Finally, I reviewed the effectiveness of spatial protection for highly migratory fish, which is related to a range of species characteristics (e.g. migration, aggregation and homing behaviors) as well as management features (fleet dynamics and management effectiveness). These results provide deeper insight into the global behavior of fishing vessels and highlight the potential and applicability of AIS vessel tracking data to document fishing and transshipment activities in unprecedented detail. By opening a new window of transparency to remote ocean areas, this work provides a foundation for improved high seas governance and management of marine living resources, especially in waters beyond national jurisdiction.

List of abbreviations used

ABBREVIATION	**DESCRIPTION**
AIS	Automatic Identification System
ANOVA	Analysis of Variance
AUC	Area Under the Curve
CBD	Convention on Biological Diversity
CCSBT	Commission for the Conservation of Southern Bluefin Tuna
CITES	Convention on International Trade in Endangered Species
CMM	Conservation and Management Measure
CMS	Convention on Migratory Species
CPUE	Catch Per Unit Effort
DM	Data Mining
EBSA	Ecologically and Biologically Sensitive Area
EEZ	Exclusive Economic Zone
ENSO	El Niño-Southern Oscillation
F	Fishing mortality (*chapter 6*)
F	Probable fishing (*chapter 2*)
FAD	Fish Aggregating Device
FAO	Food and Agriculture Organization
$\mathbf{F_{MSY}}$	Maximum sustainable rate of fishing mortality
FTP	First Passage Time (algorithm)
GDP	Gross Domestic Product
GFW	Global Fishing Watch
GPS	Global Positioning System
GT	Gross tons
HMM	Hidden Markov Model
i, j	Features (*chapter 2*)

IATTC	Inter-American Tropical Tuna Commission
ICCAT	International Commission for the Conservation of Atlantic Tunas
IMO	International Maritime Organization
IOTC	Indian Ocean Tuna Commission
ITU	International Telecommunication Union
IUCN	International Union for Conservation of Nature
IUU	Illegal, Unreported, Unregulated (fishing)
K	Number of segments used as algorithm input (*chapter 2*)
km	Kilometers
kn	Knots
LSMPA	Large-Scale Marine Protected Area
m	Meters
MMSI	Maritime Mobile Service Identity
MNM	Marine National Monument
MP	Marine Park
MPA	Marine Protected Area
MR	Marine Reserve
MSY	Maximum Sustainable Yield
mt	Metric ton
n	Number of features (*chapter 2*)
NF	Probable non-fishing
nm	Nautical miles
r	Radius
RFMO	Regional Fisheries Management Organization
S	Hidden state (*chapter 2*)
SAR	Synthetic-Aperture Radar
SEAFO	South East Atlantic Fisheries Organization
SOLAS	International Convention for the Safety of Life at Sea

SSB	Spawning Stock Biomass
$\boldsymbol{T}$	Last speed observed (*chapter 2*)
$\boldsymbol{t}$	Time (*chapter 2*)
TAC	Total Allowable Catch
UD	Utilization Distribution (algorithm)
UNCLOS	United Nations Convention on the Law of the Sea
UNFSA	United Nations Fish Stock Agreement
UTC	Coordinated Universal Time
VIIRS	Visible Infrared Imaging Radiometer Suite
VMS	Vessel Monitoring System
WCPFC	Western Central Pacific Fisheries Commission
$\boldsymbol{w_{i,j}}$	Spatial weight between features (*chapter 2*)
$\boldsymbol{x_j}$	Attribute value for features (*chapter 2*)
$\boldsymbol{Y}$	Observed variable (*chapter 2*)

Acknowledgements

“It takes a village to write a dissertation” – or, in my case, a group of people across several continents. This work is not the product of just one person, but rather the result of the guidance, help, inspiration, and support from many people. I’d therefore like to express my heartfelt gratitude and thanks to everybody who helped me to come this far:

My biggest thanks go to my main supervisor, Boris Worm. I am grateful for his incredible support, patience, guidance, and humor over the years, through ups and lows. Thank you for teaching me the art of workarounds, helping me see the big picture of things, and sharing your talent for writing, and communicating science. No amount of chocolate is big enough to repay for this.

Furthermore, I’d like to thank my co-supervisor Derek Tittensor for his help, support, and open ear to my complaints about spatial statistics. My committee members, Heike Lotze and Rodolphe Devillers, have been very helpful and supportive and I especially appreciate the many insights they shared with me. I further thank Heike and Dr. Fredda for their emotional support.

Further thanks go to all my wonderful co-authors I had the joy and privilege to work with over the past few years (and hopefully will in the future). I am amazed by the wonderful scientists out there sharing passions, dreams, edits, and data with me, and taking the time to entertain my questions.

This journey had two starting points: the first was a boat trip on a tiny Scottish island named Eigg. I’d accordingly like to acknowledge Scotland, islands (Scottish and otherwise), whisky, and boats. And John Chester, who showed me the basking sharks and the beauty of Eigg but failed to teach me how to recognize bird songs.

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Woods in a beer garden in West Virginia. My special thanks go out to both of them as well as to David Kroodsma and the rest of the teams of SkyTruth and Global Fishing Watch for sharing with me the joys and sorrows of AIS data and vessel tracking and being great collaborators and friends at the same time.

My thanks also go out to all who supported me on this journey – in no particular order: Google and Google Earth Outreach for providing (the best ever) cookies, workshops, and financial support, the TOSST program for community, teaching, and support, my awesome lab mates and all the undergrad volunteers who helped me with my work over the years, Adriene for helping me to keep my mental and physical health on track, and both Tommy and Freigeist for providing ocean time and keeping me afloat over the five years it took to complete this work. And then of course special thanks to my friends all over the world, for love, laughter, an open ear for my complaints, diverse helpful forms of beer and spirits, and support of all kinds.

Lastly, and most importantly, to my beloved family. To my partner and my grandparents for support and endless supplies of food and German sweets, and my wonderful parents, for all they have done, and are doing for me. This is for you. With love.

CHAPTER 1
INTRODUCTION

State of global fisheries

Since the first records of human fishing activity for pelagic species about 42,000 years ago (O'Connor et al. 2011) the face of global fisheries has changed significantly. Fishing vessels have progressed from small man- and wind-powered wooden boats fishing close to shore with handlines and nets, to large, ocean-going, machine-powered vessels capable of circumnavigating the globe and using sophisticated technology to find and extract about 90 to 120 million tons of fish every year (Jackson et al. 2012, Watson et al. 2013, Pauly & Zeller 2016, FAO 2018). The footprint of modern fisheries is stretching further than ever, covering more than half of the global ocean (Kroodsma et al. 2018) and reaching remote regions and depths exceeding 2,000 m (Pauly et al. 2002, Morato et al. 2006). Fisheries play a vital role in the global economy and food security, supplying 17% of the human population with a significant share of their animal protein intake and, combined with aquaculture, meet an ever growing demand for fish around the world: Per capita fish consumption has more than doubled over the last 50 years, an increase exceeding that of meat consumption from all terrestrial animals combined, and shows no signs of decreasing in the near future (FAO 2018).

The industrialization of fisheries has sped up and intensified overexploitation on a global scale (Swartz et al. 2010). While overfishing is no modern phenomenon (Jackson et al. 2001, Lotze 2007), now over 33% of all globally assessed stocks are overfished and a further 60% are maximally sustainably (fully) fished, whereas the amount of underfished stocks has declined steadily to about 7% (FAO 2018). Catches have been mostly stagnating since the mid-1990s (FAO 2018) and possibly declining when accounting for illegal, unreported, and unregulated (IUU) catches (Pauly & Zeller 2016), despite increasing effort (Fig. 1.1).

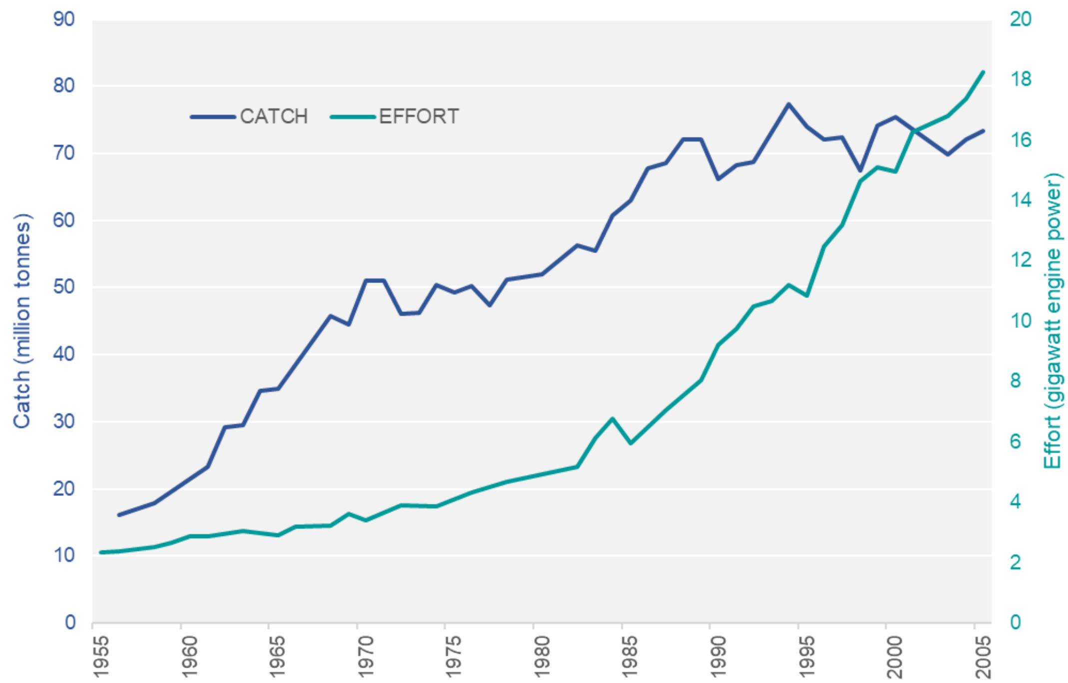

Fig. 1.1 Trends of global wild marine fish catch and fishing effort over five decades. Graph modified after data in Watson et al. 2013

Overfishing is reducing available seafood supply by an estimated 16.5 million tons per year (Ye et al. 2013). Next to overexploitation, destructive fishing practices such as bottom trawling, and illegal fishing are putting additional strain on fish stocks and marine ecosystems, raising concerns about the sustainability of global fisheries (Agnew et al. 2009).

Biological effects of fisheries

Fisheries are a dominant and widespread anthropogenic threat to marine species and ecosystems (Jackson et al. 2001, Chuenpagdee et al. 2003, Worm & Lenihan 2013), contributing directly and indirectly to habitat degradation and loss (Lotze et al. 2006), species extinction (Dulvy et al. 2003), as well as changes in food webs, and community and stock structures (Baum & Worm 2009, Lotze et al. 2011). Direct influences of fisheries include reductions of abundance and biomass of targeted species (Lotze et al. 2006, Worm

& Tittensor 2011), as well as life-history changes such as size and age at maturity (Kuparinen et al. 2016). Similar effects can occur for non-targeted bycatch species (Worm & Lenihan 2013). Indirect effects of fisheries encompass habitat destruction through destructive fishing techniques such as trawling and dredging (Thrush & Dayton 2002), as well as changes in species interactions such as trophic cascades (Baum & Worm 2009) and trophic level and community changes from long-lived, slow-growing, late-maturing species to species with shorter lifespans, faster growth rates and earlier maturity (Hutchings & Baum 2005).

Given these effects, the sustainability of many of the world's fisheries, evident for example in a constantly growing proportion of overfished stocks (FAO 2018), has been questioned (Pauly et al. 2002). Transforming fisheries into a more sustainable enterprise at a global scale will require a range of management and conservation measures (Beddington et al. 2007, Worm et al. 2009), which in turn require an adequate understanding of where, when and how fisheries are operating.

Monitoring of global fishing fleets

Unfortunately, much basic knowledge about global fisheries is vague or lacking such as the exact number of fishing vessels on the global ocean. The best estimate by the Food and Agriculture Organization (FAO) assumes around 4.6 million fishing vessels of all sizes and types fishing around the world in 2014, with about 61% under motor (FAO 2018). The largest fleets are located in Asia (79.9% of motorized fishing vessels) and mainly consist of vessels smaller than 12 m length, a further factor complicating their monitoring.

Existing information on fishing vessel activities are often rather patchy, highly aggregated, or on coarse scales. While individual government agencies in many countries monitor fisheries within national waters through vessel monitoring systems (VMS) and on-board observers, these data are generally not accessible, and not integrated at a global scale. Likewise, data for the High Seas are collected only for selected fish stocks within the convention areas of individual Regional Fisheries Management Organizations (RFMOs). In times of increased monitoring and surveillance, including the ability to track the movements of trucks, cargo ships, planes and teenage drivers, the lack of a comprehensive global overview of fishing activities is remarkable. This is presumably largely due to two factors: first, the vastness and remoteness of the realm fishing vessels are operating in complicates tracking, as the global ocean covers more than 70% of the planet's surface. Likely influenced by this, the concept of *mare liberum*, the freedom of the seas (a term coined by Hugo Grotius in the 17th century) and the open access right for navigation and fisheries, is strong within the fishing sector (Russ & Zeller 2003). This out-of-sight, out-of-mind mentality complicates monitoring, surveillance, and regulatory attempts (Rosenberg 2003, Di Lorenzo et al. 2016).

The current widespread lack of information on where, when, how much and which fishing vessels operate is causing serious problems for fisheries management, enforcement, and marine spatial planning, such as fisheries closures and marine protected areas (MPAs). Improved monitoring of fisheries has been identified as a vital part of future management efforts (Pauly & Zeller 2016).

Assessing MPA effectiveness

To encounter the rising anthropogenic pressures such as fishing, but also habitat destruction, pollution, and other stressors, international marine conservation targets such as the Convention on Biological Diversity's (CBD) Aichi target 11 were developed. Spurred by protection targets of at least 10% of the global ocean to be spatially protected by 2020. an increasing number of MPAs are being established worldwide (Lubchenco & Grorud-Colvert 2015, UNEP-WCMC & IUCN 2016). Following a definition by the International Union for Conservation of Nature (IUCN) an MPA (sometimes also called marine reserve, sanctuary, or park) is a clearly defined geographical area which is dedicated and managed through legal and other means with the aim to protect and conserve nature and associated ecosystem services as well as cultural values (Dudley 2008). Specifically excluded are areas without stated conservation goals, such as areas primarily managed for fisheries, tourism, or other industries (Day et al. 2012).

By 2018, more than 15,000 MPAs had been created globally (UNEP-WCMC and IUCN (2018), Marine Protected Planet), with sizes varying from less than a square kilometer (median size approx. 2.5 km^2) to nearly 5 million km^2, located from coastal to remote offshore waters (O'Leary et al. 2018). Over the last decade, most marine spatial protection has been achieved through the creation of large-scale MPAs (LSMPAs) covering 100,000 km^2 and more. By 2018, LSMPAs encompassed more than two-thirds of the global marine protected area and about 7% of the world's ocean. Taken together, about 7.26% of the ocean is protected (July 2018, UNEP-WCMC and IUCN (2018) Marine Protected Planet), but given levels of protection vary strongly among MPAs and only about 3.6% is included in fully implemented MPAs and even less (2%) in fully protected areas (Sala et al. 2018). The IUCN lists six different categories of protection, from category *Ia*, a strict nature reserve with limited access and activities, to category *VI*, a protected area where sustainable use of natural resources is allowed (Dudley 2008). Essential for biodiversity conservation, objectives of MPAs can include species-specific management and stock rebuilding, protection, maintenance or restoration of marine ecosystems, and their processes, services, and associated species, and other specific targets (Day et al. 2012). While not typically intended as a tool in fisheries management, MPAs have been recognized as a vital part of ecosystem-based management, contributing to protecting and rebuilding stocks and ecosystems affected by fisheries (Roberts et al. 2005, Gaines et al. 2010).

Developing a new monitoring tool

New tools to study and understand the influence of fisheries and marine management have become available fairly recently. While VMS has been used for many years by individual governments and RFMOs to monitor fishing vessels in their respective area, these systems are typically for government use only, and data are hard to access, coarse, and often highly aggregated (Lambert et al. 2012, Russo et al. 2016). Over the past years, the Automatic Identification System (AIS) has increasingly being used for monitoring and research of vessel activities (Natale et al. 2015, de Souza et al. 2016, McCauley et al. 2016, Wu et al. 2017). In contrast to VMS, AIS is a global open-access system with data provided by multiple suppliers worldwide. AIS transponders are mandated as a safety feature to avoid collisions for large (>300 gross tons) fishing vessels undertaking international voyages (SOLAS Convention Chapter V) but are also used by smaller ships. Vessel locations as well as identity, speed, course over ground, and a variety of other information are transmitted to land-based towers as well as receivers on low-orbit satellites as frequently as every couple of seconds (Fig. 1.2) and can be used to map a vessels' tracks and analyze its behavior based on movement patterns.

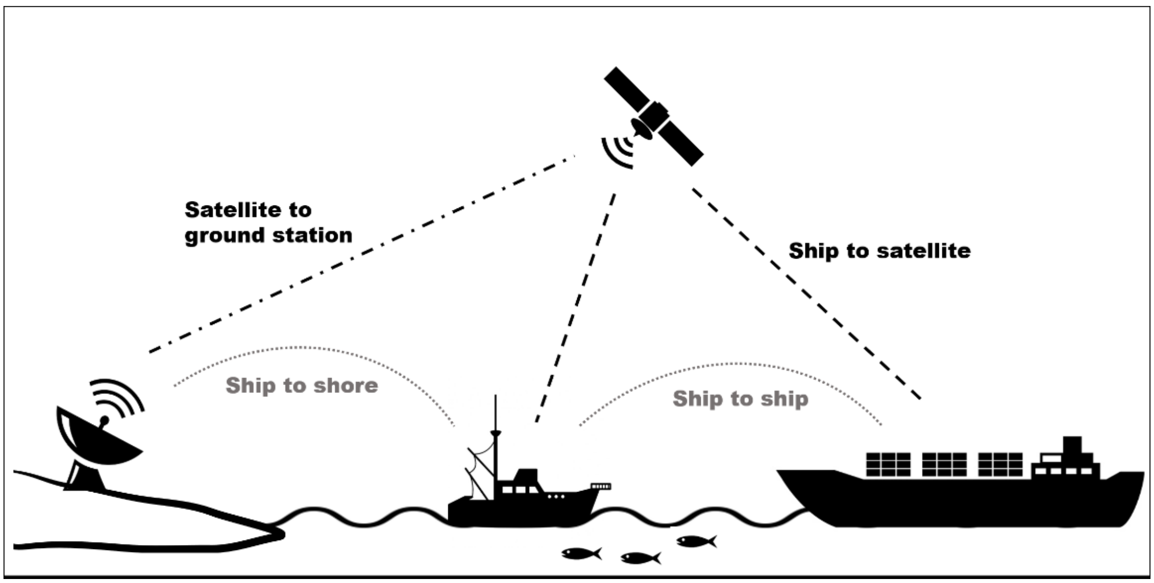

Fig. 1.2 Scheme of Automatic Identification System (AIS) signaling between ships carrying the transponder as well as satellite and ground stations.

AIS data are greatly extending the range of hitherto existing systems such as VMS and observer systems and enable a range of new research questions and management possibilities. Using AIS tracks and analysis tools based on machine learning techniques, it is now possible to detect, classify and map the spatial and temporal patterns of global fishing vessel movements (Kroodsma et al. 2018) and relate them to areas of interest such as sensitive habitats and MPAs as well as understand the links of fishing vessels to international seafood supply chains even for vast and remote areas.

Therefore, the overarching goal of my thesis is to explore the application of global fishing vessel monitoring in the context of marine conservation and fisheries management more broadly. I ask

1) if it is possible to track and analyze the behavior of fishing vessels in relation to MPAs,
2) how spatial and temporal patterns of fishing effort compare around various MPAs and
3) how this knowledge can inform MPA and fisheries management.

Extending on that, I examine the usefulness of vessel tracking data for other issues such as the transshipment of catch at sea and review how spatial protection and management can be optimized for highly mobile species.

Structure of thesis

My thesis is divided into five chapters exploring different aspects of the analysis and application of AIS data to track and map the behavior of fishing vessels, and a final concluding chapter highlighting main findings, applications and next steps. In Chapter 2 I present novel techniques to determine fishing effort for three different fishing gear types (trawl, longline, and purse seine) from AIS data. Building on this in Chapter 3, I analyze purse seine fishing effort around the iconic Galápagos Marine Reserve, one of the oldest LSMPAs in the world. In Chapter 4, I extend the scope and investigate fishing effort around thirteen LSMPAs worldwide. Exploring additional applications of AIS data in the context of marine conservation, I examine the role of transshipment of catch from fishing vessels to refrigerated cargo vessels in Chapter 5 and analyze how this affects seafood supply chain transparency and traceability. Finally, in Chapter 6 I survey the literature whether spatial protection is feasible and beneficial specifically for large pelagic fishes such as tuna and sharks and their associated fisheries, before drawing conclusions from my work and highlighting future research in Chapter 7.

Ultimately, my work aims to contribute to an increased comprehension of the role and scope of global fisheries and novel tools to track, understand, and eventually regulate their effects on marine ecosystems, and their living resources.

Statement of Co-Authorship

This dissertation contains five data chapters. Each chapter corresponds to a manuscript written for publication in a scientific journal and largely follows the regular structure of scientific papers consisting of an abstract, introduction, materials and methods, results, discussion, and conclusion.

All co-authors contributed to these manuscripts through comments, advice, support in research design and method development, as well as interpretation. The publication status of each chapter at the time of submission of this thesis is as follows:

Chapter 2: de Souza, E.N*., **Boerder, K.***, Matwin, S. and Worm, B., 2016. Improving fishing pattern detection from satellite AIS using data mining and machine learning. PloS ONE 11, e0158248.

* equal co-authors

Chapter 3: **Boerder, K.**, Bryndum-Buchholz, A., & Worm, B., 2017. Interactions of Tuna Fisheries with the Galápagos Marine Reserve. Marine Ecology Progress Series 585, 1–15

Chapter 4: Interactions between large marine protected areas and global fishing fleets (*unpublished*)

Chapter 5: **Boerder, K.**, Miller, N.A., Worm, B., 2018. Global hot spots of transshipment of fish catch at sea. Science Advances 4, eaat7159

Chapter 6: **Boerder, K.**, Schiller, L., Worm, B. Not all who wander are lost: spatial protection for large pelagic fishes. Marine Policy (*in revision*)

Four of these chapters (2, 3, 5 and 6) have either been published or submitted. Details are provided on the first page of each chapter. Chapter 2, 3 and 5 have been published under an open-access license.

Data accessibility

Data is freely available through globalfishingwatch.org and upon request to research@globalfishingwatch.org and kristina.boerder@dal.ca.

CHAPTER 2
Improving fishing pattern detection from satellite AIS using data mining and machine learning

Abstract

A key challenge in contemporary ecology and conservation is the accurate tracking of the spatial distribution of various human impacts, such as fishing. While coastal fisheries in national waters are closely monitored in some countries, existing maps of fishing effort elsewhere are fraught with uncertainty, especially in remote areas and the High Seas. Better understanding of the behavior of the global fishing fleets is required in order to prioritize and enforce fisheries management and conservation measures worldwide. Satellite-based Automatic Information Systems (S-AIS) are now commonly installed on most ocean-going vessels and have been proposed as a novel tool to explore the movements of fishing fleets in near real time. Here we present approaches to identify fishing activity from S-AIS data for three dominant fishing gear types: trawl, longline and purse seine. Using a large dataset containing worldwide fishing vessel tracks from 2011 - 2015, we developed three methods to detect and map fishing activities: for trawlers we produced a Hidden Markov Model (HMM) using vessel speed as observation variable. For longliners we have designed a Data Mining (DM) approach using an algorithm inspired from studies on animal movement. For purse seiners a multi-layered filtering strategy based on vessel speed and operation time was implemented. Validation against expert-labeled datasets showed average detection accuracies of 83% for trawler and longliner, and 97% for purse seiner. Our study represents the first comprehensive approach to detect and identify potential fishing behavior for three major gear types operating on a global scale. We hope that this work will enable new efforts to assess the spatial and temporal

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de Souza, E.N., Boerder, K., Matwin, S. and Worm, B., 2016. Improving fishing pattern detection from satellite AIS using data mining and machine learning. PloS one, 11(7)

distribution of global fishing effort and make global fisheries activities transparent to ocean scientists, managers and the public.

Introduction

A common challenge in ecology is the mapping of dynamic patterns of human activity across vast areas in order to understand and track their ecosystem impacts on regional and global scales (Halpern et al. 2008, Trebilco et al. 2011, Selig et al. 2014). While important from a scientific perspective, there are also many other obvious applications, including the monitoring of marine fisheries and the enforcement of spatial management measures, such as marine protected areas (MPAs), ecologically and biologically sensitive areas (EBSAs) as well as fisheries closure zones. While the reception range of coastal monitoring tools such as tower-based applications (tAIS, radar) is limited to inshore areas, long-range tools such as AIS (Automatic Identification System) and VMS (Vessel Monitoring System) provide insight into vessel movements elsewhere. Vessel monitoring systems were specifically designed to monitor commercial fisheries while AIS was intended as a safety feature to avoid vessel collisions under low visibility. While the use of VMS devices is mandated only for some fleets in individual nations, the International Maritime Organization (IMO) has made the carrying of an AIS transponder mandatory for all vessels larger than 300 gross tons or carrying passengers (SOLAS Chapter V). In addition, national regulations may include other vessel types, such as per recent requirements by the European Union that all fishing vessels bigger than 15 m must carry an AIS device (Natale et al. 2015). Both VMS and AIS feature on-board transmitters linked to the vessel's GPS to receive and transmit exact position in time and space on long-range radio frequencies to either coastal ground stations or satellites. In the case of AIS, data are also transmitted to other ships in the area that carry the device. VMS usually transmits in time intervals varying from one to several hours, satellite-based AIS (S-AIS) transmissions can be as frequent as every few seconds, enabling the monitoring of fine-scale vessel behavior and movement patterns. Several attempts have been made to use VMS and AIS data to understand fishing vessel behavior, for example by using simple presence/absence or vessel speed (Gerritsen & Lordan 2011, Chang & Yuan 2014). While speed can be a useful indicator of vessel activity, operational speeds while fishing vary greatly for different fishing gear types such as trawls, longlines or nets. More sophisticated algorithms differentiating fishing from non-fishing activity for different fleets and gears are needed to properly capture and represent the characteristics of the various fishing methods, as stated previously by Natale et al. (2015). We develop and present such algorithms here, then assess their accuracy in correctly identifying individual fishing events or 'sets' by comparing against expert-labeled data. Finally, we briefly chart potential applications in marine ecology, conservation, and fisheries management.

Materials and Methods

Data Sets

This work is based on a database containing global AIS data obtained from AIS-enabled communication satellites since January 2011 until October 2015. Data were obtained under research license from exactEarth (http://www.exactearth.com/products/exactais). A representation of three several-year tracks and examples of fishing activity patterns for trawling, longlining and purse seining is given in Fig. 2.1. Individual tracks for known trawl, longliner and purse seine fishing vessels were extracted from different regions representing vessels from different nations operating in various parts of the oceans at scales from coastal fishing grounds to circumnavigating the globe. The trawler data contained an initial sample of 83 vessels operating in the North Pacific and corresponds to 217,860 data points collected in July 2013 used for algorithm development and training. For comparison and testing on a global scale, a second trawler data set, composed of seven vessels operating from January 2011 until October 2015 across various ocean basins was selected. These tracks were much longer than those in the North Pacific, totaling 884,478 data points. Analyses for longliners comprised data from 16 vessels operating across all major ocean basins from June 2012 until December 2013, corresponding to

573,204 data points. Data on seven purse seine vessels comprised 399,545 points from January 2011 until October 2015 representing long-range operations in various areas of the world.

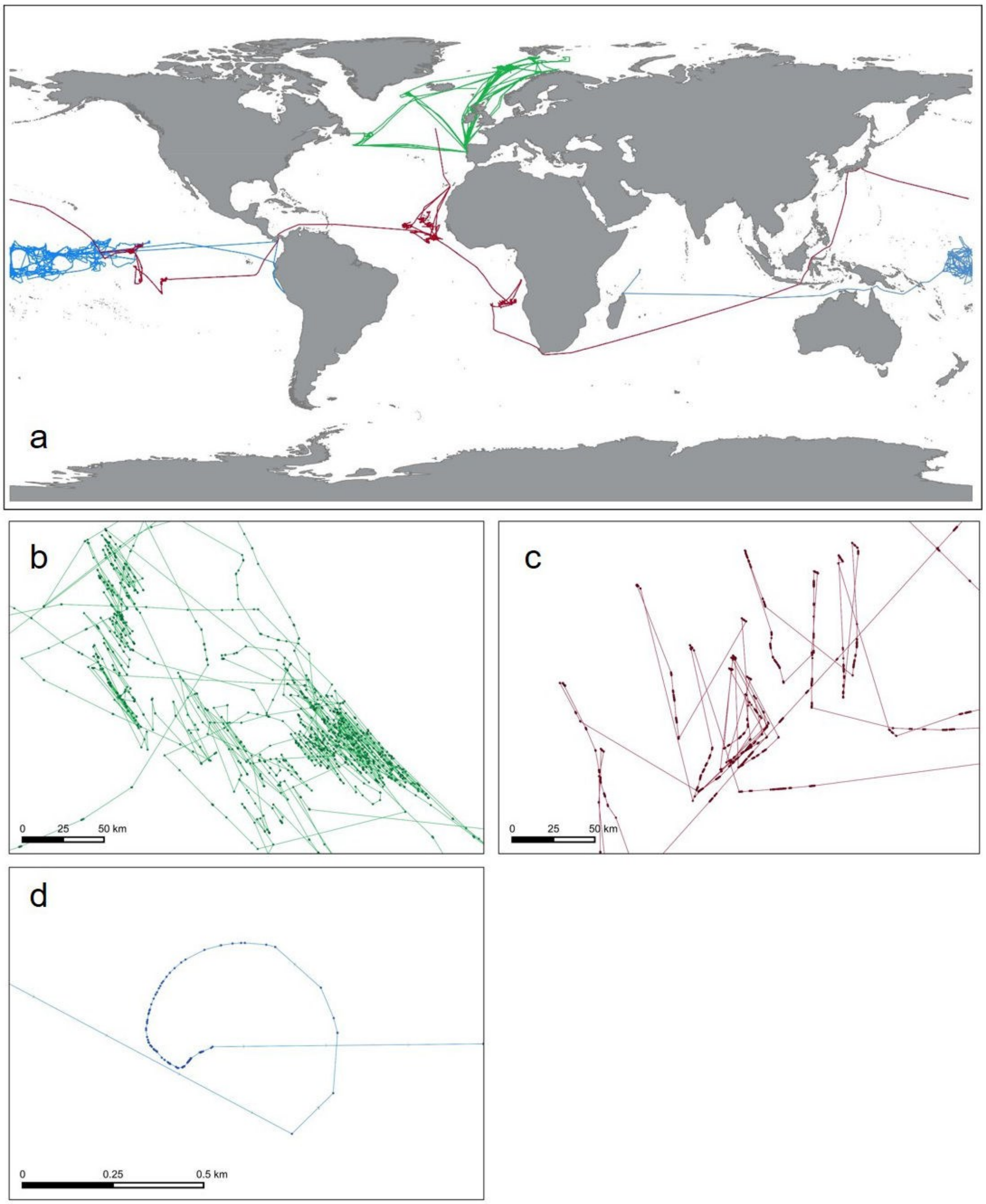

Fig. 2.1 Presentation of raw AIS tracks for three individual vessels using different fishing gear types. Global overview (A) and more fine-scale representations of potential fishing behavior for a trawler (green, B), longliner (red, C) and purse seiner (blue, D). Dots represent individual AIS signal detections, lines interpolated tracks. Note the global-range behavior of longliners, and the more regional basin-wide operations of purse seine and trawl vessels. Map data by Natural Earth.

Definition of Fishing Activity by Gear Type

Trawler

Trawling involves dragging one or more nets behind a fishing vessel either on the sea floor (bottom trawling) or in the water column (pelagic or midwater trawling). While trawling, fishing vessels usually slow down and aim to maintain a constant speed to keep the strain on the dragged net as even as possible. Duration of trawling operations depends mostly on the density of the prey and can last from a few minutes up to several hours. The typical length of a trawl will vary between 3 and 5 hours (FAO). Here, trawling activity is defined from the moment the net is deployed to when it is retrieved. Trawls are often characterized by slow, steady speeds between 2.5 and 5.5 knots. These speed thresholds were determined directly from the distribution of the AIS speed data and correspond to similar values obtained from literature (Lee et al. 2010, Skaar et al. 2011, Alemany et al. 2013, Mazzarella et al. 2014).

Longliner

Longlining involves the setting of fishing lines (up to 100 km length) equipped with several hundred to several thousands of hooks (FAO). Lines can be deployed at various depths with the use of floats and horizontal lines extending to deeper waters. To set the line, the vessel travels only slightly slower than its steaming speed while the line is set. After the last hook is in the water, the line is left in the water for some hours ('soak time'). During this time, the vessel either drifts slowly with the line or sets other lines in the vicinity. To haul the line the vessel reverses and steams back along the line. The whole operation can take up to a day. Speed while hauling is kept mostly constant but can vary according to catch and number of crew working. The time to set a longline depends on the length of the line and the number of crew working it, but the median set time estimated from the 16 vessels used in our analyses was 6.5 hours. Here a longline set is considered to start with setting of the longline and to end with retrieval of the last hooks. Characteristics used for identification of longline sets comprise spatial-temporal movement patterns in a very restricted area.

Purse seiner

Purse seines are long nets deployed hanging vertically from floats around schooling fish on or near the surface by the vessel or by a separate skiff. To avoid fish escaping the setting of the seine needs to happen quickly and is done at high speeds averaging around 10 knots. Once the net encircles the school completely, the bottom of the net is pulled shut and the net hauled. Drifting with the net attached, the fish are then retrieved and transferred to the vessel. The duration of this process depends on the amount of catch and can vary from one to several hours (Walker & Bez 2010). For the purpose of this work, a purse seine set is defined as the time the net is closed around the fish to the end of the fish bailing operation when the net is lifted out of the water. During this time the purse seine vessel stays more or less stationary and speed over ground is generally slow, ranging around 2.5 knots and less. This threshold was determined based on speed distributions of the AIS data as well as observations from literature (Bertrand et al. 2005, Bez et al. 2011).

Data Labeling and Pre-processing

All vessel tracks were classified and pre-labeled as potential fishing and non-fishing events by an expert based on information on fisheries characteristics as obtained from literature, analyses of the tracks (speed and movement profiles by gear type, flag, vessel size and area of operation), personal interviews with fishermen and fisheries on-board observers and comparisons to speed and movement profiles from observer data for the Northeast Atlantic. Characteristics include speed over ground, change of direction within a defined area, spatial-temporal movement patterns, operational time and duration of the fishing event. The testing of the algorithms against expert-labeled data was chosen because suitable observer or logbook data for the fleets and time period examined were unavailable to us. Expert judgement on vessel behavior based on the aforementioned characteristics might be a conservative approach, as some fishing events will be missed. In order to improve fishing activity prediction, for each data point we calculated whether it occurred during night or day. In order to estimate the amount of sun light available in a region of the world during a certain UTC-based time, the R package solaR (Perpiñán Lamigueiro 2012) was used, with positive values for sunlight marking the day, zeroes marking the night. To avoid irregular vessel movement patterns very close to shore and in port a 10 km boundary around shorelines was established. The calculation required the computation of the Haversine distance between each vessel track point and all the points in the shoreline data provided by Natural Earth to establish the minimum distance to shore. This process is computationally expensive and to improve the quality of the calculation, it was decided to use the parallel capabilities provided by the code available under https://stackoverflow.com/questions/27697504/ocean-latitude-longitude-point-distance-from-shore.

Algorithm Testing

The algorithms presented here were tested against expert-labeled fishing vessel tracks, separating fishing and non-fishing activity based on observations from operational data, expert knowledge and comparison to other tracking data. Accuracies presented are based on these comparisons. Each algorithm proposed has different assumptions: The Hidden Markov Model (HMM) assumes that the user will have part of the data available for training, while the Data Mining (DM) and filtering approaches do not require training. These differences determine how these approaches are tested: the HMM algorithm (applied for trawlers) uses Monte Carlo experiments to measure how it behaves with time variation, and the DM approach uses all data available for testing against expert-labeled data (used for longliner and purse seiner). For the trawling activity the HMM algorithm was tested with a Monte Carlo simulation using the implementation provided by the R package DMwR (Torgo 2003). The North Pacific trawler data set was used for this purpose. This data was the first data set that was available for development and testing of the HMM approach and offered a high variation of vessel behaviors within shorter tracks covering one month of data (July 2013). To ensure the applicability of the HMM trained on this data set, the HMM was subsequently applied and tested on a second data set containing multi-year trawler tracks operating in various parts of the ocean on regional and global scale. The Monte Carlo Simulation partitioned the data in 20 segments; each of these segments was trained with anywhere from 25,000 data to 130,000 points. All tests predicted 100,000 points in the future. The Monte Carlo simulations did not consider the 10 km threshold, as the HMM uses only speed as input. We decided to execute tests with fixed-size windows for training and test to avoid possible overfitting. The DMwR package randomly selects 20 data windows according to the user specification for training and test data, where the entire window is dislocated once the training and testing is done, and its respective statistical results are stored. The results of the testing are represented as Prediction (F for fishing) and Prediction (NF for non-fishing) in Table 2.1 and Table 2.2 and were calculated based on Altman & Bland 1994. These two metrics give an estimate of how well the algorithm predicts the desired class using unseen data.

Table 2.1 Performance measures for the worldwide trawl dataset. NF stands for probable non-fishing and F for probable fishing events. Sensitivity is related with non-fishing detection, and specificity with fishing detection. The column Stat. Diff. Fish Effort shows the t-test statistical comparison (*p*-value) between the predicted fishing effort time calculated from the algorithm's labels and the expert's labels. The asterisk indicates a significant difference.

Track ID	**Track Size**	**Accuracy**	**Prediction (F)**	**Prediction (NF)**	**Sensitivity**	**Specificity**	**% of Fish. Activity**	**Stat. Diff. Fish. Effort**
1	38,258	0.75	0.47	0.96	0.90	0.71	0.42	0.84
2	254,323	0.84	0.83	0.85	0.93	0.69	0.70	0.21
3	93,670	0.83	0.82	0.95	0.99	0.40	0.89	0.11
4	56,287	0.87	0.89	0.80	0.94	0.69	0.77	0.04*
5	55,034	0.92	0.51	0.98	0.82	0.93	0.14	0.21
6	285,407	0.57	0.28	0.93	0.84	0.51	0.55	0.09
7	101,499	0.76	0.01	1.00	1.00	0.76	0.24	0.32
Median±SD		0.83±0.11	0.51±0.32	0.95±0.07	0.93±0.07	0.68±0.17		

Table 2.2 Performance measures for the 16 longliner vessels in different oceans. NF stands for probable non-fishing and F for probable fishing events. Sensitivity is related with non-fishing detection, and specificity with fishing detection. The column Stat. Diff. Fish Effort shows the t-test statistical comparison (p-value) between the predicted fishing effort time calculated from the algorithm's labels and the expert's labels. Two of the vessels could not be measured because they did not have any labeled fishing activity. The asterisk indicates a significant difference.

Track ID	Track Size	Accuracy	Prediction (F)	Prediction (NF)	Sensitivity	Specificity	% of Fish. Activity	Stat. Diff. Fish. Effort
1	7,935	0.46	0.25	0.95	0.35	0.93	0.70	0.29
2	25,558	0.80	0.87	0.56	0.52	0.88	0.80	0.08
3	9,642	0.65	0.57	0.85	0.45	0.91	0.71	0.62
4	35,258	0.89	0.93	0.70	0.71	0.93	0.81	0.67
5	34,993	0.87	0.89	0.82	0.66	0.95	0.79	0.28
6	42,566	0.89	0.90	0.83	0.54	0.98	0.88	0.35
7	25,287	0.76	0.73	0.81	0.59	0.89	0.68	0.67
8	96,314	0.54	0.48	0.97	0.22	0.99	0.87	0.05*
9	123,686	0.89	0.90	0.89	0.81	0.94	0.67	0.54
10	128,668	0.74	0.66	0.96	0.49	0.98	0.75	0.82
11	2,070	0.54	0.50	0.81	0.20	0.94	0.86	0.80
12	1,452	0.95		0.95	1.00	0.00	0.00	NA
13	12,405	0.71	0.43	0.93	0.68	0.82	0.44	0.34
14	18,169	0.86	0.87	0.80	0.55	0.96	0.83	0.67
15	6,421	0.99		0.99	1.00	0.00	0.00	NA
16	2,780	0.88	0.80	0.93	0.89	0.86	0.36	0.30
Median±SD		0.83±0.15	0.87±0.11	0.57±0.24	0.77±0.21	0.93±0.04		

Data Analysis

HMM and Observation Variable Choice for Trawlers

Hidden Markov Models (HMM) represent a probability distribution over a sequence of points (Ghahramani 2001). It is assumed that an observation at time t was generated by a hidden state S_t. The second assumption is that given the value in S_{t-1}, the value of S_t is independent of all previous states to $t-1$. Assuming that the observed variable is defined by Y_t in time t, and states S_t are defined as classes {'F','N'}, then the Markov Model is factored in the following way:

$$\Pr(S_{1:T}|Y_{1:T}) = \Pr(S_1)\Pr(Y_1|S_1)\prod_{t=2}^{T}\Pr(Y_t|S_t)\,Pr(S_t|S_{t-1}) \quad (1)$$

Where $Pr(S_t|S_{t-1})$ represents the transition matrix giving the probability of a state being changed to another state. In the case of fishing activity, it will represent the probability of changing the vessels' state from fishing to non-fishing, and vice-versa. This transition matrix can be estimated directly from the distribution of fishing and non-fishing labels defined in the data set. $\Pr(Y_t|S_t)$ represents the probability of an observed variable occurring associated with a state S_t in time t. T represents the last speed read in the data set.

In order to build a successful HMM model it is necessary to define which parameters offer the best chance to identify the correct hidden states (fishing or non-fishing). Since speed is a key feature in all ecological work associated with HMM's of animal movement (Peel et al. 2011), our work also implements an HMM based on speed.

Fig. 2.2 shows the speed distribution for vessels engaged in potential fishing or non-fishing activities (such as steaming, searching and anchoring). These results are comparable to those presented by others for mobile towed gear (Charles et al. 2014).

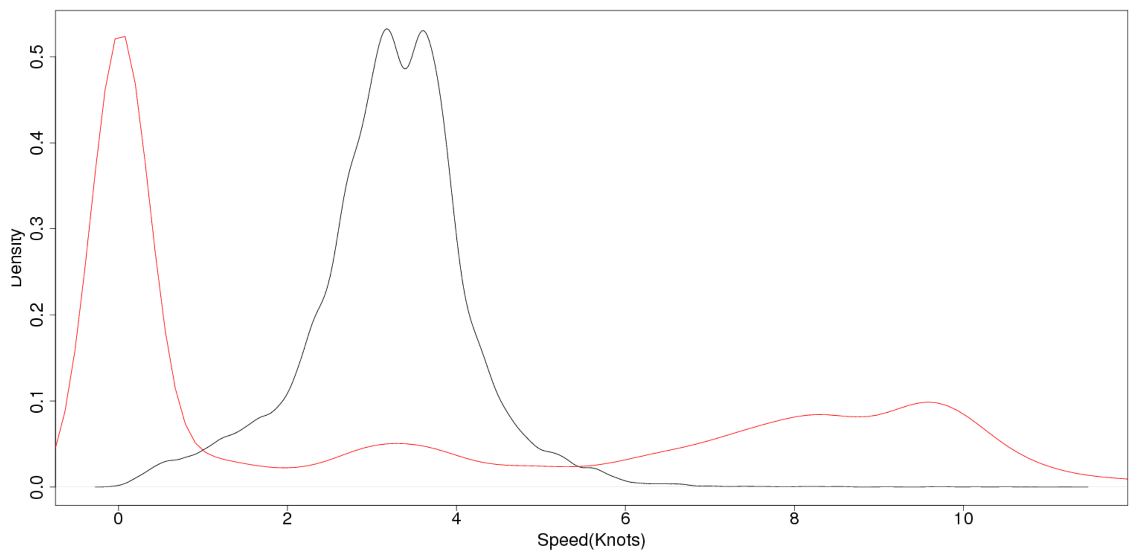

Fig. 2.2 Speed distribution for trawlers during fishing and non-fishing activities. The red line represents probable non-fishing activity, and the black line probable fishing activity.

Data Mining Approach for Fishing Detection of Longliner

The same HMM approach cannot be directly applied to the longliner data set as the speed distribution does not follow a clear pattern as seen in the trawler data (Fig. 2.3). Several parameters contained in the data were tested as potential classifiers, but none proved sufficient to describe fishing and non-fishing activity patterns for longlining. Therefore, we opted to develop an alternative approach similar to what biologists have used in studying animal movement tracks. It has been shown that human fishermen tend to show similar movement patterns as animal predators simply because this is the most efficient method to search for and locate prey (Bertrand et al. 2005). Building on this, we decided to use a segmentation technique traditionally applied to animal predators. The Lavielle's segmentation algorithm (Lavielle 1999, 2005) is widely used by biologists to segment animal tracks in order to identify possible variations in their habitat use.

The Lavielle's algorithm finds the best segmentation of a time series assuming that it is built by K segments defined by the user. The algorithm is not originally designed to work with GPS coordinates, but instead it will segment *any* regular time series data. Before segmentation the whole track was regularized in time to remove gaps in the GPS readings. The regularization assumes equal separation of seven hours between each GPS reading. The separation of seven hours was defined based on the average time that longliners spent fishing as estimated from the labeled data set.

Lavielle's algorithm searches for a minimum contrast estimator in a problem of change points estimation, which suggests that Lavielle's algorithm is a drift detector in a time series. The implementation used required the definition of *K*, which is the number of segments desired by the user as algorithm input, and it was estimated as 70 segments based on various tests. More information about the implementation used may be found in the adehabitat package (Calenge 2006) in R.

For each of the segments a second algorithm was used to detect if the segment is composed of straight or curved lines. For each segment, the algorithm calculates the cosine of the relative angle between two consecutive points and calculates an average for all points in the segment. If the average cosine returns a value above 0.8 or below -0.8, it indicates that the whole segment is formed by a straight-line movement. All segments that contain the majority of their points classified as straight lines receive a label of non-fishing activity. The segments presenting curved movements are separated to execute a third algorithm to further filter non-fishing activity.

Once the curved segments are separated it is possible to analyze each point to remove possible non-fishing activity classified as fishing. Since the entire curved segment is considered fishing from the straight-line algorithm detection, many non-fishing activities will be automatically assumed as fishing. In order to reduce this type of error, two other algorithms are combined to extract these false alarms: First-Passage Time algorithm (FPT) (Fauchald & Tveraa 2003) and Utilization Distribution algorithm (UD) (Worton 2018).

The FPT algorithm (Johnson et al. 1992, Fauchald & Tveraa 2003) uses Brownian Motion theory to find areas where the patterns appear in a trajectory. According to Calenge (2006), "for a given scale *r* it is defined as the time required by animals to pass through a circle of radius *r*". This means that the FPT algorithm searches for the minimum radius r that contains multiple passes of the animal/vessel in the same region. One problem with the approach is that, depending on how long the track is, it does not restrict the size of *r*, which could result in a longer search for the correct radius. Fortunately, the movement of the vessels inside the curved segments is very restricted, which makes the search for *r* feasible. After some tests, we found that *r* using 30 different radii uniformly distributed varying from 0.1 to 1, offer an acceptable accuracy for FPT. Fauchald and Tveraa (2003) extended the FPT algorithm to compute the variance of the *log(FPT)*, which should be high for scales where vessels have multiple passes. As a threshold for our algorithm, if the *Var(log(FPT))* ≤ 0.1 it is a straight line and these points are labeled as non-fishing.

To reduce possible false alarms, the Utilization Distribution algorithm (UD) (Worton 2018) was used in addition. The UD is defined as a probability distribution (Van Winkle 1975) using only the longitude and latitude features. In order to estimate this distribution, a kernel method clustering algorithm is used in the coordinate parameters. The idea is to use a bivariate kernel function as a distance metric in each GPS location to find the cluster centers. The adehabitat implementation uses by default the normal kernel function, and we did not change this parameter. This work uses the UD estimations to correct wrong predictions of points wrongly classified as fishing activity to non-fishing activity.

The combination of FPT and UD within the curved data segments offers an extra 1% to 2% accuracy improvement and a reduction of non-fishing activity false alarm comparing to the expert labels.

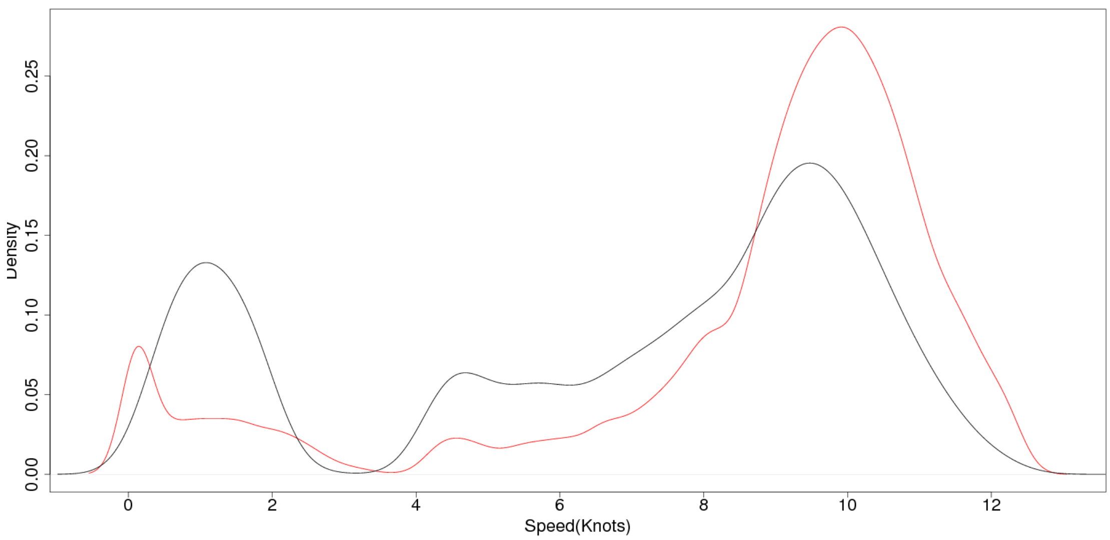

Fig. 2.3 Speed distribution for longliners during fishing and non-fishing activities. The red line represents probable non-fishing activity, and the black line probable fishing activity.

Filtering Approach for Purse Seiner Fishing Detection

Fishing activity detection for purse seiners builds on two assumptions based on literature data (see Bez et al. 2011), personal communication with fisheries observers working on board of various purse seiners and observations from the AIS data. Firstly, the majority of purse seiners do not fish at night, with some exceptions that are not considered here. Second, that the fishing pattern consists of two main activity patterns, namely the setting of the net at high speeds and the drifting while hauling in the net and retrieving the fish at very low speeds. While the setting of the net is a very short activity that may not be represented in the data due to insufficient satellite coverage, the hauling and bailing can take up to several hours and is thus used to detect and classify potential fishing activity. Using the abovementioned day/night classifier, possible fishing activity was detected using a speed filter for speeds smaller than or equal to 2.5 knots. Fig. 2.4 presents the speed distribution for purse seiners, for all positions reported at least 10 km from shore and during day. It is noticeable that the majority of probable fishing activity happens with speeds in the range of 0 up to 5 knots, as indicated by the black distribution, and a second distribution peak appears for probable non-fishing activity (around 15 knots) in red.

The 2.5 knots speed threshold was chosen based on observations from the AIS data and the work of Bez et al. (2011). The filtering-based approach does not require machine learning, but the computation of extra features as described in the data pre-processing section.

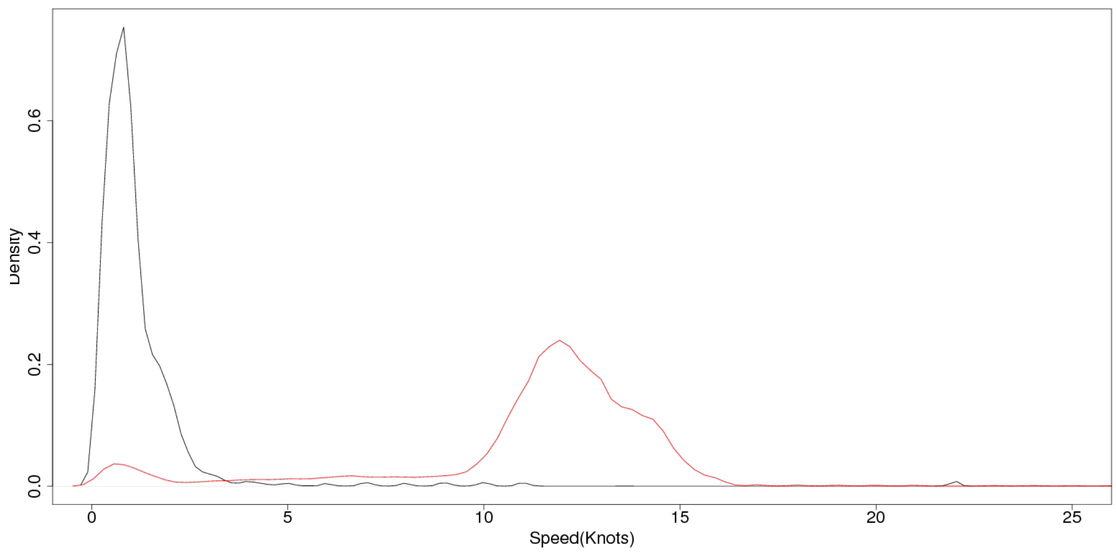

Fig. 2.4 Speed distribution for purse seiners. The red line indicates the probable non-fishing activity labeled by the expert, while the black line represents probable fishing activity. This distribution considers only the speeds reported by the vessels more than 10 km from shore and during day time

Results

Trawler

As presented before, the solution proposed for trawling vessels is based on the HMM algorithm. As HMMs assume that the data is time dependent the analysis must consider the order of the points. Monte Carlo simulations are the only repeatable testing method that does not change this order. The objective of the repeated test is twofold: 1) to assess the HMM performance, and 2) to identify how many data points are required to correctly predict potential fishing activity.

Fig. 2.5 presents the Monte Carlo average results of the accuracy, recall and Area Under the Curve (AUC) for each class (fishing and not fishing). The HMM keeps the same average accuracy independently on the number of points available for training (close to 88%). The algorithm also improved the recall (defined as the percentage of relevant, correct fishing detections retrieved by the algorithm) of the fishing activity from 58% to above 85% when the algorithm was presented with more data for training, but also reduced the prediction of non-fishing activity from above an average 95% to 87% at the same time. This increase in the number of correct predictions of probable fishing activity indicates that the HMM may be overfitting, with higher number of points available for training. The overfitting aspect of the model can only be evaluated with tests on a different data set.

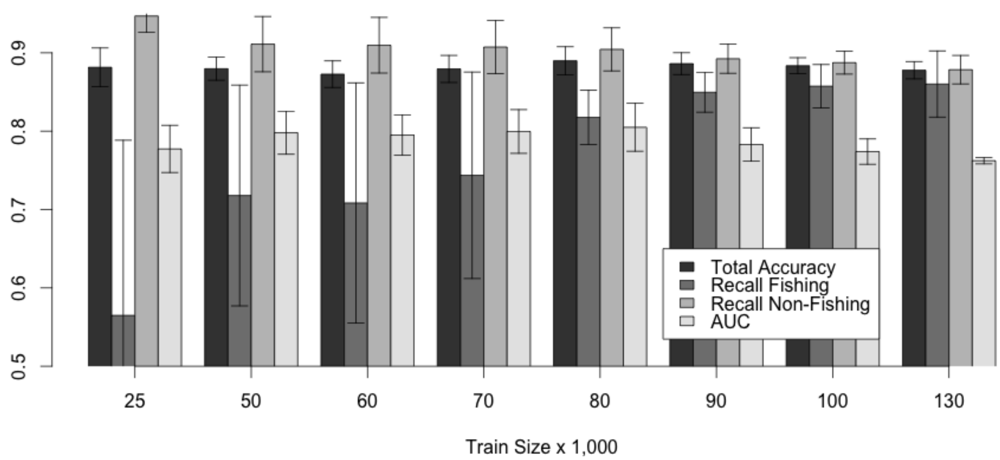

Fig. 2.5 Accuracy/Recall measured for trawlers with a Hidden Markov Model using a Monte Carlo Simulation. Results do not consider the 10 km threshold.

Despite the fact that the Monte Carlo simulations are a good indicator of the algorithm's performance, they do not give the full information about how the algorithm will work with future data sets. To confirm the number of points necessary for training we tested the HMM against data from other parts of the world. The training was done on the 2013 North Pacific data, which contains 25,000 points. The test data was derived from the seven vessels operating across all oceans and years. The algorithm showed low accuracy results when trained with more than 25,000 data points, due to overfitting. This lies in the nature of the Monte Carlo test methodology: the more data available for training also means less data for testing, which reduces the variance (Fig. 2.4). To avoid overfitting, it is advisable to train the HMM with fewer data points to increase its generalization. The results indicate that the HMM can be used to locate probable fishing activity for trawlers using tracks from different areas of the world with a median accuracy of 84%. There was little difference in accuracy for the two subtypes of trawling, pelagic/midwater (average accuracy 75%) and bottom (average accuracy 80.6%). The sensitivity to detect probable fishing activity (column Sensitivity in Table 2.1) shows a median of 93% and the respective specificity (the capability to identify probable non-fishing activity) a median of 68%.

Fig. 2.6 presents the results for track number two from Table 2.1, containing 254,323 points with a total accuracy of 84% and 69% specificity to detect probable fishing activity, as well 93% sensitivity of probable non-fishing activity detection. As shown by Altman & Bland (1994), the sensitivity and specificity only concern the current model's capability to classify the test instances, but they give no information about future algorithm performance. With support of the information provided in columns Prediction (F) and Prediction (NF), it is possible to confirm that the algorithm will have high probability to keep the same performance with unseen data.

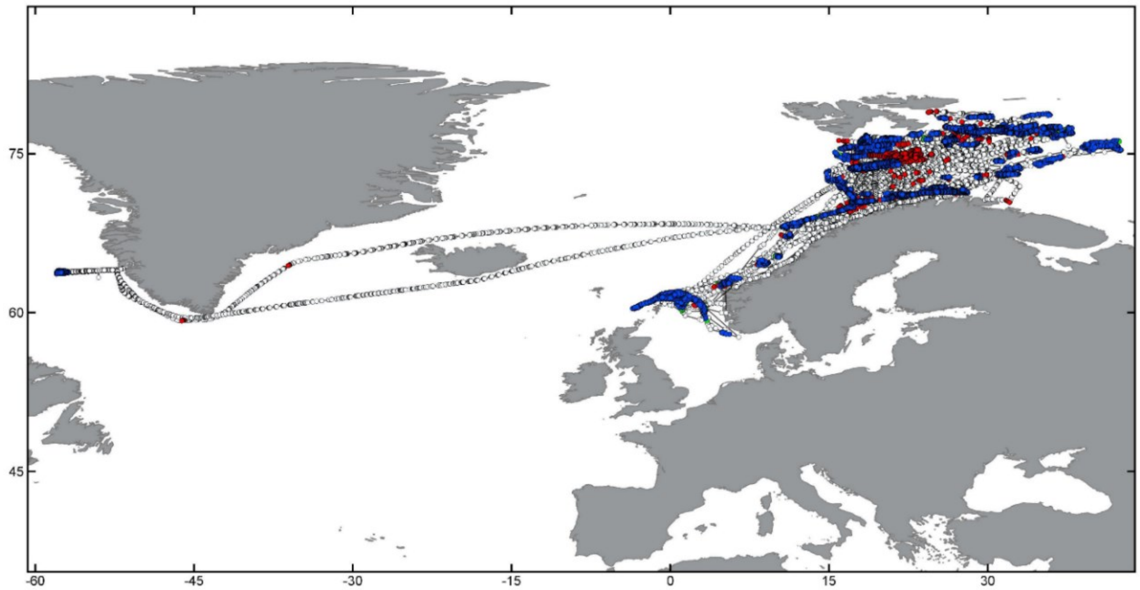

Fig. 2.6 Comparison of the Hidden Markov Model algorithm results to the expert labels for trawler number 2 in Table 2.1. Matching results for fishing activity presented in blue, expert labels in green and the algorithm's fishing activity predictions in red. Empty circles represent non-fishing activity as identified by algorithm and expert. Map data by Natural Earth.

Longliner

The 16 longliner vessels were tested independently as a mathematical model was fit to the data set and all the data was used for testing. The longliner database contains an average of 76% of movement patterns dedicated to assumed fishing activity. Table 2.2 summarizes the results of the longliner detection. In general, the median algorithm performance is 83%. As previously presented, the columns Prediction (F) and Prediction (NF) shown in Table 2.2 indicate how well the algorithm will perform on unknown data sets.

Fig. 2.7 and Fig. 2.8 illustrate the results of the fishing prediction for tracks 8 and 1 in Table 2.2 respectively. These tracks were chosen because these vessels presented the highest and lowest accuracy results in the longliner data.

The last column presented in Table 2.1 and Table 2.2 shows the results of a t-test between the algorithm's predicted Fishing Effort (FE) and the expert's labels. The FE is a measure to estimate how much time the vessel invests in fishing activity and is calculated in two steps: 1) periods of fishing activity for each vessel are tagged at the moments when a change from non-fishing activity to fishing and vice-versa occurred, and 2) for each individual period of fishing the time difference between the first and last AIS messages is calculated. A statistical difference between expert and prediction labels ($p < 0.05$) occurred only in one case ($p = 0.05$, Table 2.1 and Table 2.2). This indicates that the algorithm is capturing the expert labeling for nearly all of the vessels evaluated. Two of the vessels were not part of the analyses as they did not have any fishing activity labeled by the expert.

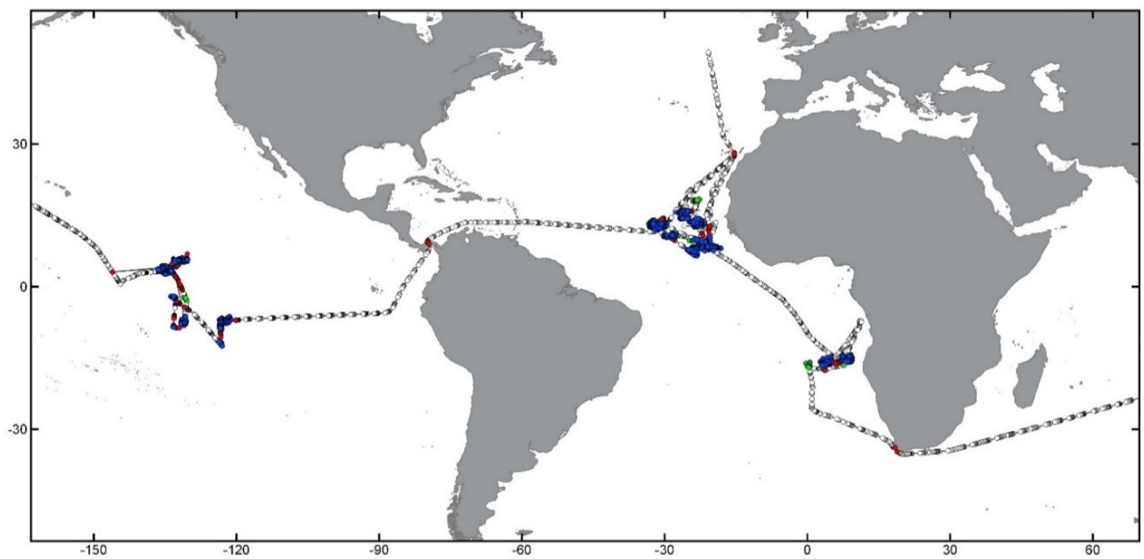

Fig. 2.7 Comparison of algorithm results to expert labels for longliner number 8 from Table 2.2 (accuracy 89%). Matching results for fishing activity presented in blue, expert labels in green, and the algorithm's fishing activity predictions in red. Empty circles represent non-fishing activity as identified by algorithm and expert. Map data by Natural Earth.

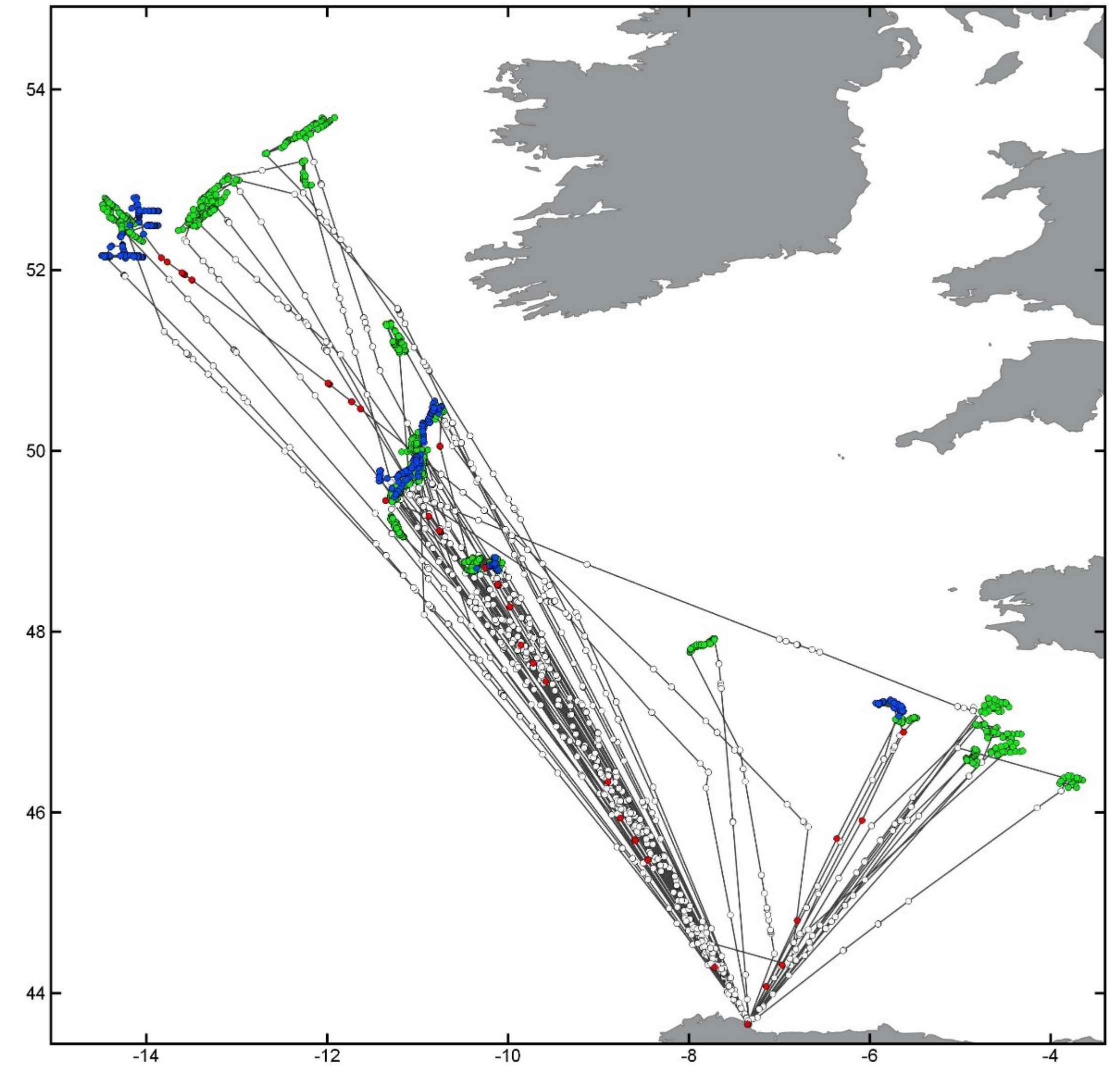

Fig. 2.8 Comparison of algorithm results to expert labels for longliner number 1 from Table 2.2 (accuracy 46%). Matching results for fishing activity presented in blue, expert labels in green, and the algorithm's fishing activity predictions in red. Empty circles represent non-fishing activity as identified by algorithm and expert. Map data by Natural Earth.

Purse Seiner

Table 2.3 illustrates the results for the Purse Seiner filtering approach. The total median accuracy of the model is 97%, with a standard deviation of 1%. The filter median prediction capability of probable fishing and non-fishing activities is 97% and 94%, respectively. The main difference is in the sensitivity measure associated with probable non-fishing activity, with a median of 99% and a standard deviation of 1%. The specificity associated with probable fishing activity detection is around 71% with a standard deviation of 17%.

All these results indicate that the model is highly accurate to detect probable fishing activity when comparing to expert labels. The FE metric was not estimated for purse seiners as due to incomplete satellite coverage the total purse seining activity from starting to set the net to the finishing of the haul is rarely seen completely. Subsequently the filtering was designed to capture the hauling portion of the fishing activity only. This makes the FE metric unreliable to estimate the correct probable fishing time intervals. Instead, we present the AUC which is a common metric used in ML to inform on algorithm performance. The median AUC for the purse seiner is 0.85, with a standard deviation of 0.08, which also indicates the model is reliable to identify the minority class. Fig. 2.9 presents an example of the results found with the filtering technique proposed and corresponds to track number six in Table 2.3. The track contains 51,545 points and the filtering algorithm reached 97% total accuracy with nearly all probable fishing activity detected. The detection of false alarms is based on the expert's labeling strategy, which classified speeds higher than 2.5 knots as fishing.

Table 2.3 Results for the purse seiner filtering approach. The seven vessels were randomly chosen from multiple parts of the world. NF stand for probable non-fishing and F for probable fishing activity. Sensitivity is related with non-fishing detection and specificity with fishing detection.

Track ID	Track Size	Accuracy	Prediction (F)	Prediction (NF)	Sensitivity	Specificity	AUC	% of Fish. Activity
1	43,457	0.95	0.02	1.00	0.95	0.82	0.89	0.05
2	170,972	0.98	0.94	0.98	1.00	0.71	0.85	0.05
3	43,369	0.96	0.95	0.96	1.00	0.36	0.68	0.02
4	18,122	0.94	0.76	0.97	0.96	0.79	0.88	0.13
5	38,596	0.99	0.91	1.00	1.00	0.89	0.94	0.03
6	51,545	0.97	0.97	0.97	1.00	0.65	0.83	0.05
7	33,484	0.97	0.96	0.97	1.00	0.64	0.82	0.05
Median±SD		0.97±0.01	0.97±0.01	0.99±0.01	0.94±0.34	0.71±0.17		

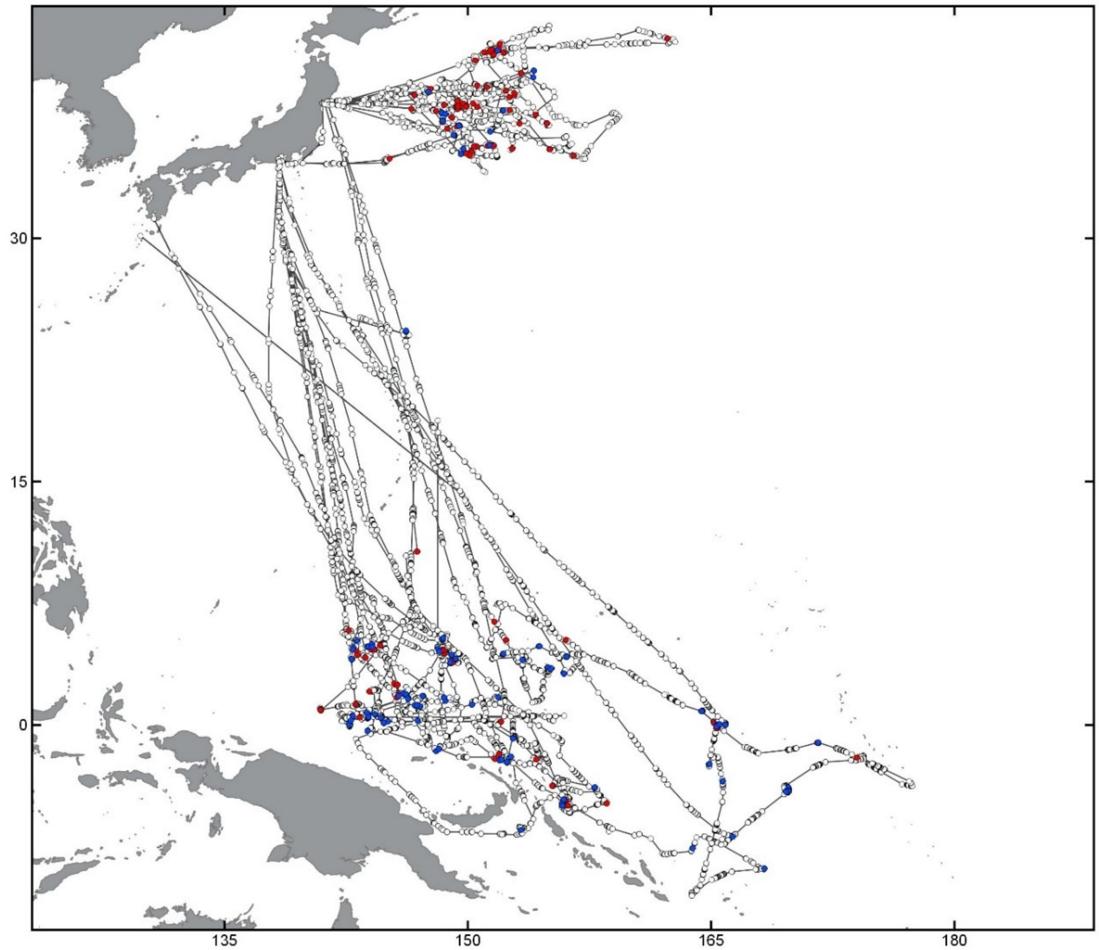

Fig. 2.9 Comparison of algorithm results to expert labels for purse seiner number 6 from Table 2.3. Matching results for fishing activity presented in blue, expert labels in green, and the algorithm's fishing activity predictions in red. Empty circles represent non-fishing activity as identified by algorithm and expert. Map data by Natural Earth.

Discussion

The aim of this work was to develop automated methods to detect potential fishing behavior from different gear types based on AIS track data. As fishing activity for each gear type has its unique pattern and characteristics (compare Fig. 2.1), we developed separate approaches tailored to the specific challenges associated with systematic differences in vessel behavior, speed distribution, and fishing time.

Results indicate that our machine learning and data mining approaches were able to correctly identify a very large fraction (83–97%) of probable fishing events compared to the expert labeled data. Importantly, our algorithms performed similarly well for different fleets in different regions operating from regional to global scales, making this a versatile tool to study the behavior of fishing fleets worldwide. While a number of previous studies have analyzed VMS data for vessel identification and tracking (Witt & Godley 2007, Vermard et al. 2010, Gerritsen & Lordan 2011), AIS data have only been available for a few years and we are aware of only two published attempts to use it for detection and classification of vessel activities (Lehner et al. 2009, Natale et al. 2015). Most previous work was done in engineering and computer science, largely either focusing on the technical aspects of system performance analyses mostly in the context of Maritime Situational Awareness (Cervera & Ginesi 2008, Cervera et al. 2011, Carson-Jackson 2012, Pallotta et al. 2013, Liu, De Souza, et al. 2014, Liu et al. 2015, Soleimani et al. 2015) or on the computational challenge of complex analyses on big data and the combinations with other tools (Cairns 2005, Lehner et al. 2009). While these studies form a valuable basis for the work presented here, no previous study has focused on a comprehensive solution to detect and classify fishing activities on a global scale using a distinction between gear types.

We presented three distinct machine learning, data mining and filtering approaches to detect potential fishing activity for trawlers, longliners and purse seiners, respectively. The method developed for trawlers is based on previous works (Vermard et al. 2010) that showed a HMM is an effective way to predict fishing activity using vessel speed as the critical parameter. Our model works equally well for the two subtypes of trawling we accounted for, pelagic/midwater and bottom trawling. To establish the performance of the trawl algorithm, a Monte Carlo experiment was executed in which the algorithm performs increasingly better when the number of data points available for training increases. The objective to run Monte Carlo simulations is to assess the minimum number of points required to train a stable model and confirm the model's performance. Unfortunately, Monte Carlo simulations can suffer from overfitting, which is the case when tested models perform very well during the simulation but fail when new data is presented. Monte Carlo simulation results presented in Fig. 2.5 indicate that models trained with 80,000 points would offer a more reliable and stable model, but when we tested these models against the data from various parts of the world, the accuracy reduced considerably. In order to confirm if the 80,000 points HMM was overfitting we tested HMM models with higher number of points for training. Results remained much below the accuracies reported by the Monte Carlo simulations, therefore confirming the overfitting assumption. As more generic models can be created with a reduced amount of data to avoid overfitting, we tested the HMM model with 25,000 points and results were very close to those reported by the Monte Carlo simulations for various parts of the world.

For longliner fishing activity detection no clear separation between fishing and non-fishing speeds exists. To detect probable longliner fishing activity, the development of an alternative algorithm was required. Since the proposed method is a Data Mining-based approach, all the data was used for testing, and the results indicate that the proposed method based on previous analyses of animal movement and habitat selection (see Bertrand et al. 2005) offers a good prediction level to detect a more complex type of fishing pattern. The main results are presented in Table 2.2 and indicate that the algorithm has good prediction capabilities with a median accuracy of 83%. One disadvantage of the method is the track segmentation algorithm, which requires defining the number of segments beforehand. Further work might consider methods to dynamically partition tracks.

Finally, the purse seiner approach uses another DM algorithm based on a filtering strategy. The filter designed is similar to the one used by McCauley et al. (2016).The algorithm filters the data assuming that purse seiners only fish during day, and that fishing activities are characterized by low speeds (lower than 2.5 knots). Due to this type of behavior, the filter results are well aligned with expert labels, indicating that the filtering approach is well suited to this type of vessel. A limitation of the filter is that it will not capture probable fishing activity with speeds above the 2.5 knots threshold and at night.

This assumption opens the possibility of future research to create techniques that are based on movement shape to detect possible fishing activity at higher speeds.

The labeling for all three methods is point-based, attaching a fishing or non-fishing label to the individual AIS vessel position records. On the basis of these labeled points, fishing time per area can be calculated on any given scale. Future work includes the partitioning of sequences of labeled points into sets for each gear type to adjust the output of the methods presented here to other units commonly used as measures of fishing effort. This can also include further information on the characteristics of fishing vessels as either transmitted with the AIS message or available through vessel registry data bases such as size, tonnage or engine power.

Overall, the algorithms are slightly more likely to detect potential fishing activity than expert labels, reflecting the conservative approach taken in labeling. As most groundtruthed data such as observer and logbook data is proprietary and often impossible to access, manual vessel activity labeling by an expert provides a workable solution. A possible next step to further improve the algorithms is to test and train them on data containing groundtruthed fishing activity recordings such as observer or logbook data if available at a resolution matching that for vessels carrying AIS transponders. This would eliminate any variability potentially introduced through the manual labeling and provide a more precise picture of vessel activities.

Despite their flexibility and versatility, the approaches presented here come with limitations and caveats. The biggest weakness is the structure of the AIS system itself: not all vessels carry AIS transponders and those who do can still tamper with or disable the transponders or falsify positional or identification data. Manipulated data and switching-off events can be detected using specially designed algorithms, but more comprehensive legislation regarding the use of AIS may be needed to address these non-compliance issues on a broader scale, e.g. as suggested by McCauley et al. (2016). Another issue regarding AIS data is the limited satellite coverage, which at this point provides limited time and space windows for observation, and samples some regions better than others.

This inherent problem, however, will be alleviated with the projected launching of several new AIS-enabled satellites in 2016.

The methods presented here were designed to stand on their own, but they perform best on pre-processed AIS data, where wrong detections, noise and faulty out-of-bounds data (e.g. observations on land) are removed. There are efforts currently underway to develop better 'despoofing' algorithms that can reliably detect faulty or falsified AIS messages with inaccurate information. In combination with such pre-processing and de-spoofing algorithms, our methods will be applied within the framework of 'Global Fishing Watch' (http://www.globalfishingwatch.org), an open-access online-tool to detect and visualize fishing activity worldwide. Furthermore, we caveat that algorithm performance varies with the quality of the track, giving better results for long tracks with high detection continuity. The three approaches presented here are designed to be applied for vessels with known gear type, so the appropriate method can be chosen. While work on the identification of gear types used by fishing vessels is ongoing, our methods are only partly applicable for vessels using mixed gears and not for other gears than the three addressed here.

Conclusion

The approaches we have developed allow us to detect and identify potential fishing behavior for three main gear types with high accuracy and spatial resolution on a global scale. This opens a new window of transparency, providing information on ocean uses not only to marine spatial planners and managers as well as the public, but also lays a foundation for future scientific research on vessel behavior for different gear types and sizes. One major challenge lies in expanding behavior identification to small-scale and artisanal fishing vessels which currently remain largely invisible in most tracking systems. Spatial-temporal analyses of long-term tracking data will offer valuable insights into fishing effort intensity and distribution in various areas such as nation's exclusive economic zones, the high seas and areas of special interest like MPAs and other areas of biological or managerial interest.

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Author Contributions

Concept of paper by KB and BW. Design of experiments by ENS, KB, SM, and BW. Data analyses and experiments by ENS and KB. Discussion and writing by KB, ENS, SM, and BW.

CHAPTER 3
Interactions of Tuna Fisheries with the Galápagos Marine Reserve

Abstract

The largest protected areas of any kind have all recently been established in the ocean. Since 2012, five protected areas that exceed one million square kilometers in size have been implemented, mostly in remote oceanic areas. The potential conservation and fisheries benefits of such reserves have been debated in the public, the media, and the scientific literature. Little is known about their effectiveness for commercially valuable pelagic predators, especially for highly migratory species such as tuna and billfishes. Here we analyze the iconic Galápagos Marine Reserve, documenting interactions with and changes in associated tuna purse seine fisheries in the Eastern Tropical Pacific. Using a combination of long-term onboard observer data and a novel data set of high-resolution, remotely sensed vessel tracks (Automatic Identification System, AIS), we reveal progressive divergence of tuna fishing effort, catch, and catch per unit of effort trends in areas adjacent to the reserve from trends in the surrounding fished areas. Both data sets show a regionally unique hot spot of concentrated effort along the western reserve boundary now receiving more than four times the fishing effort density than the rest of the surrounding area. These dynamic interactions of tuna purse seine fisheries with the Galápagos Marine Reserve suggest that the reserve might enhance fish stock availability to local fisheries and help to stabilize local catches despite overall decreasing biomass trends for these highly commercial tuna stocks.

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Introduction

Growing concerns about the state of marine biodiversity and its resources have provided incentive for the creation of large-scale marine protected areas (MPAs), which have increased greatly in number and size in recent years (Lubchenco & Grorud-Colvert 2015). A protected area, according to the definition by IUCN, is a clearly defined geographical space, recognized, dedicated, and managed, through legal or other effective means, to achieve the long-term conservation of nature with associated ecosystem services and cultural values (Day et al. 2012). Since 2012 alone, five MPAs larger than 1,000,000 km^2 have been created and collectively protect an area that is significantly larger than all MPAs gazetted in the previous 125 years. Just in 2016, the expansion of the Papahānaumokuākea Marine National Monument in Hawaii followed by the declaration of the Ross Sea Marine Protected Area off Antarctica created the world's largest protected areas of any kind, land or sea, covering more than 1.5 million km^2 each. Declarations of further large MPAs such as the Rapa Nui Rahui MPA around the Easter Islands, Chile, followed in 2017. This recent trend has sparked a discussion about the benefits and challenges of large scale protection for key stakeholders such as fisheries (Wilhelm et al. 2014). While some conservation benefits of large MPAs have been addressed in previous studies (e.g. White et al. 2017), the role of large MPAs for fisheries management remains unclear especially in the context of pelagic fisheries operating outside of reserve boundaries (Roberts et al. 2001, Forcada et al. 2008). It has been suggested that large MPAs can be an effective conservation tool, even for some pelagic species (Edgar et al. 2014), particularly when placed around vulnerable aggregation sites such as nursery or spawning grounds (Gell & Roberts 2003). While there are studies showing benefits of marine reserves for some small-scale fisheries in coastal areas, these mostly concern resident species such as lobster (Kelly et al. 2000, Follesa et al. 2011), scallops (Murawski et al. 2000), clams (Tawake et al. 2001) or reef fishes (McClanahan & Mangi 2000, Stobart et al. 2009, Da Silva et al. 2015, Tewfik et al. 2017). Effects of MPAs or other spatial closures on highly mobile predators such as tuna or billfish are more elusive (Jensen et al. 2010, Grüss et al. 2011, Edgar et al. 2014). Due to the migratory nature of these oceanic species, they are within reach of open-ocean fisheries for most of their lives, potentially contributing to documented declines (Myers & Worm 2003, Juan-Jordá et al. 2011, Pons et al. 2017), range contractions (Worm & Tittensor 2011), and heightened extinction risk for some long-lived high-value species (Collette et al. 2011). Marine reserves are often thought to be too small to provide sufficient range coverage (Hyrenbach et al. 2000, Breen et al. 2015) and consequently, limited benefits for highly migratory species and associated fisheries.

Understanding the interactions between large MPAs and marine fisheries could significantly advance our comprehension of MPA effectiveness as a conservation and management tool (Horta e Costa et al. 2013, Stevenson et al. 2013). On the one hand, MPAs that disallow fishing might simply displace effort to other areas that may then be subject to overfishing (Halpern et al. 2004). On the other hand, and particularly if an area has been heavily fished prior to protection, MPA establishment could initiate rebuilding of resident stocks that might then lead to spillover, here defined as the net emigration of organisms across the reserve boundary (Buxton et al. 2014). Over time, increasing spillover of rebounding stocks may also increase catches in adjacent fishing grounds (Gell & Roberts 2003, Murawski et al. 2005). Fishermen tend to capitalize on this phenomenon by fishing along reserve boundaries, a behavior known as 'fishing the line' (Rijnsdorp et al. 1998, Johnson et al. 1999, Russ & Alcala 2011, Alemany et al. 2013, Van Der Lee et al. 2013). This pattern has been documented in several theoretical studies but empirical evidence is mostly available for small-scale marine reserves or temporal fisheries closures (Murawski et al. 2005, Kellner et al. 2007, Goñi et al. 2008, Stobart et al. 2009, Van Der Lee et al. 2013).

Anecdotal evidence from the Galápagos archipelago suggests that industrial purse seine vessels are 'fishing the line' for tuna around the Galápagos Marine Reserve, and that Ecuadorian tuna fishermen are changing their fishing habits and are now supporting the reserve as a perceived tuna spawning and nursery area (Kliffen & Berkes 2015). These recent developments follow a long history of tuna fishing in Ecuador and around the Galápagos islands that began in the 1920s with commercial purse seiners from California and later Japan targeting mainly yellowfin (*Thunnus albacares*), bigeye (*Thunnus obesus*) and skipjack tuna (*Katsuwonus pelamis*). Following legislative changes and Ecuador's claiming of the 200-nautical mile Exclusive Economic Zone (EEZ), tuna fisheries became more restricted. However, tuna was still caught around the islands by purse seiners and from the 1960s on by longliners, until the passage of the Special Law in 1998 banning commercial fishing within the Galápagos Marine Reserve (Oxford et al. 2009). This was mainly motivated by increasing conflicts between commercial purse seine fishing vessels and the growing tourism industry as well as local fishermen (Kliffen & Berkes 2015). In recent years the fishery has been dominated by vessels flagged to Ecuador and Mexico as well as a number of other South American countries with about 272 active purse seiners licensed by the Inter-American Tropical Tuna Commission (IATTC) for the Eastern Tropical Pacific. Stock assessment models estimate that overfishing is not occurring for the three main tuna species with reported catches close to maximum sustainable yield (yellowfin; Minte-Vera et al. 2014) or below (bigeye and skipjack; Maunder 2014; Aires-da-Silva & Maunder 2014). With the establishment of the Galápagos Marine Reserve in 1998, industrial tuna fisheries were banned within a 40 nm radius around the Galápagos islands. This was only the second large MPA to be declared worldwide (after the Great Barrier Reef Marine Reserve in Australia). It provides a valuable case study as it meets at least four of the five key so-called NEOLI features (*sensu* Edgar et al. 2014) that best predict protected area effectiveness worldwide; specifically no-take regulations (industrialized fisheries are banned inside the Galápagos Marine Reserve, artisanal fisheries are allowed but not currently targeting tuna), age (18 years of protection), large size (about 133,000 km^2), and isolation from the mainland (Edgar et al. 2014). The fifth key feature (enforcement) was weak at the time of reserve implementation in 1998, but improved after 2002 (Castrejón & Charles 2013, Kliffen & Berkes 2015). Examining interactions between the Galápagos Marine Reserve and associated tuna purse seine fishery prior and subsequent to reserve enforcement represents an unparalleled opportunity to empirically document changes in tuna fisheries associated with a large-scale MPA.

Considering qualitative reports of increasing support for the Galápagos Marine Reserve among tuna fishermen (Kliffen & Berkes 2015) we attempted to quantify interactions between the Galápagos Marine Reserve and associated tuna fisheries from two complementary data sources: detailed catch and effort data including data on the use of fish aggregating devices (FADs) from the Inter-American Tropical Tuna Commission on-board observer program (henceforth referred to as IATTC observer data) and satellite-based vessel position data, delivered by Automatic Identification Systems (AIS) for purse seine vessels in the wider area around the Galápagos Marine Reserve (Fig. 3.1). We hypothesized that purse seine fisheries adjacent to the Galápagos Marine Reserve will exhibit different trajectories subsequent to reserve establishment when compared to surrounding waters, due to interactions with the reserve.

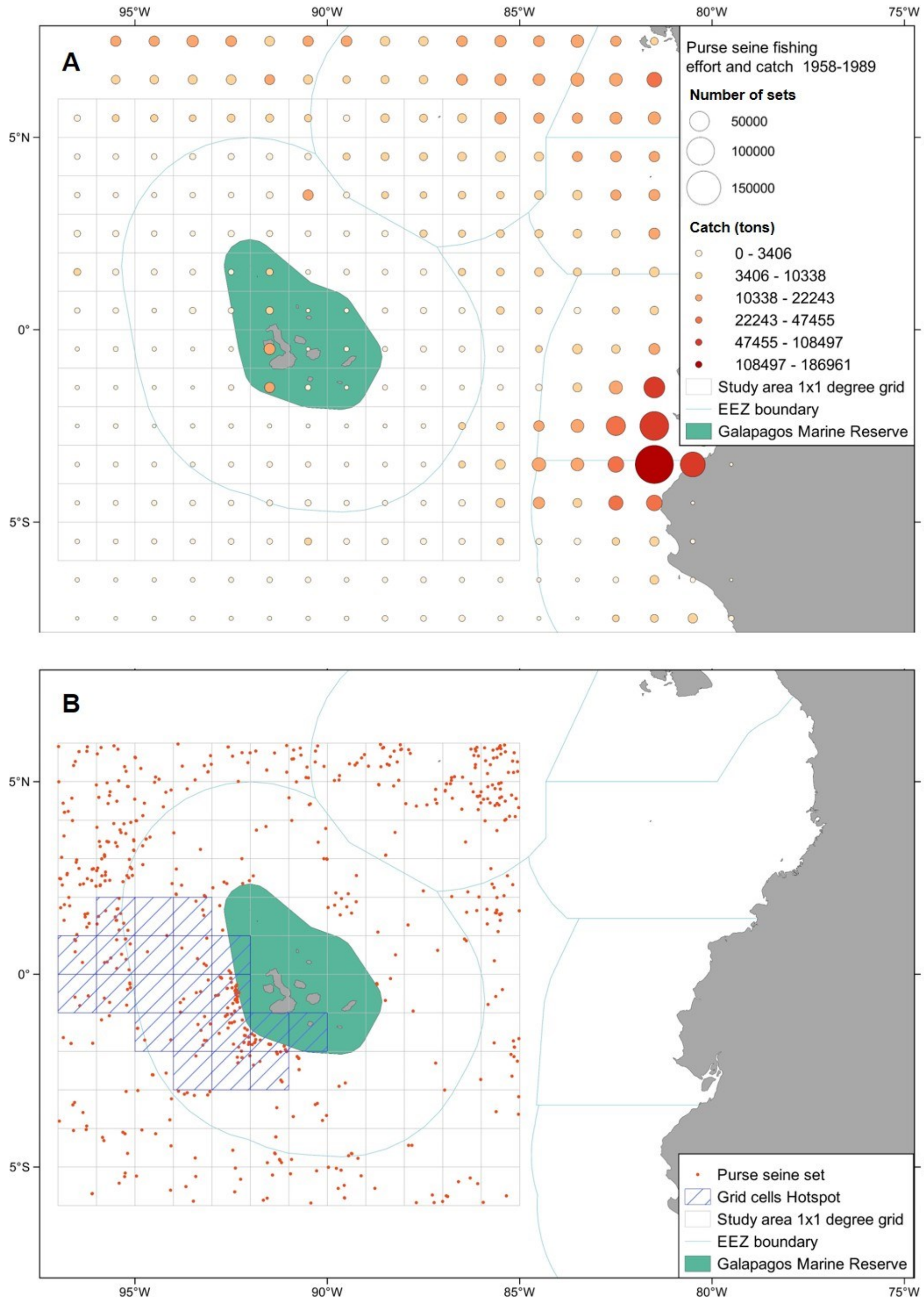

Fig. 3.1 Tuna fisheries in the study area around the Galápagos Islands, Eastern Tropical Pacific. Shown is the marine area around the Galápagos Islands with the current marine reserve highlighted in green and exclusive economic zones outlined in light blue. (**A**) Historical fishing effort and catch prior to the study period and reserve establishment (1958-1989), (**B**) Positions of purse seine fishing sets identified from AIS data for 49 vessels operating within the study area from January 2011 to October 2015 (red dots). Note the aggregation of sets along the western marine reserve boundary and around the western Galápagos Exclusive Economic Zone (a fishery 'hot spot' is indicated by hatched area, see data in Fig. 3.2). Map data from Natural Earth and MarineRegions.org, and IUCN and UNEP-WCMC, The World Database on Protected Areas (WDPA), May 2016.

Materials and Methods

Study area

We selected a large bounding box between 97°W, 6°S, 85°W and 6°N around the Galápagos Islands as our study area, which includes the Galápagos Marine Reserve and the majority of the Ecuadorian EEZ around the islands totaling more than 1.7 million km^2. This area was subdivided into 1x1° cells, corresponding to the aggregation of the IATTC observer data used in this study (Fig. 3.1). For further study, a hot spot comprising 21 cells was determined west and southwest of the Galápagos Marine Reserve based on highest intensity fishing effort as seen from IATTC and AIS data (Fig. 3.1 and Fig. 3.2).

Data Sources

IATTC observer, FAD, and stock assessment data

Scientific fisheries observer data were obtained from IATTC for the tuna purse seine fishery (vessel with > 363 mt carrying capacity) in the Eastern Tropical Pacific for the years 1990-2015. Only fishing trips with observers on board are included in the dataset, which represent 86% of the total fishing effort in the tuna purse seine fishery within the respective vessel capacity during the study period. The data set contains information for each purse seine set, aggregated by month and grid cell. Each record includes year, month, and the coordinates for the respective grid cell location of the set, the number of sets, and the total catch by species (catch and discards of tuna and non-tuna species). Catch-per-unit-effort (CPUE) was calculated as the total catch per cell per year divided by the number of fishing sets in that cell and year. Raw data by year are displayed in the Supplement (Fig. A.1).

Data on the use of fish aggregating devices (FADs) was obtained from IATTC for the years 1990 to 2015. This data set includes information on number of sets per area, month and year as well as set types and total catch by species.

These data provide a coarse-resolution view of spatial and temporal patterns in observed fishing effort and catch. To compare broader trends in catch rates with those in recruitment and biomass, published stock assessments for principal tuna species as published by IATTC were accessed and digitized. These data provide assessment-model based estimates of annual recruitment and spawning stock biomass ratios relative to the estimated virgin biomass that would exist without fishing.

AIS data

The Automatic Identification System (AIS) is a maritime safety tool intended to prevent ship collisions. Since 2004 AIS transponders are mandatory for all fishing vessels larger than 300 gross tonnage (GT) on international voyages (International Maritime Organization – Safety of Life at Sea (SOLAS) Convention Chapter V, Annex 17). The AIS transponder transmits position and ship identification data at regular intervals to surrounding ships carrying receivers. Signals are also picked up by ground stations carrying receivers and by AIS-equipped satellites in whose field of view the ship is located. The receiving stations transmit the data to a processing center. A complete transmitted AIS message contains, amongst other information, the ship's Maritime Mobile Service Identity (MMSI), International Maritime Organization (IMO) number, call sign, speed and course over ground, position, rate of turn and possibly the destination, the ship name and the type of vessel. With this high-resolution data the fine-scale movements of each vessel carrying an AIS transponder can be visualized (de Souza et al. 2016, McCauley et al. 2016).

Data for each vessel present in the study area between January 2011 and October 2015 with more than 200 AIS detections were extracted from a database containing AIS data from the commercial provider exactEarth (http://www.exactearth.com). Through comparison with international fleet registries (International Maritime Organization Global Integrated Shipping Information System (GISIS) *https://gisis.imo.org*) and regional fisheries management organization vessel register lists (IATTC Regional Vessel Register https://www.iattc.org/VesselRegister/VesselList.aspx?List=RegVessels&Lang=ENG) using MMSI and IMO numbers we identified all purse seine fishing vessels active in the study area for further analysis.

Data Analyses

IATTC observer data

Spatial autocorrelation in the observer data time series was tested for using Moran's I test for spatial autocorrelation as used in the *spdep*-package in the R statistical computing environment (R Version 1.0.136). Spatial clustering of fishing effort, catch, and CPUE was analyzed conducting a hot spot analysis based on the Getis-Ord G_i* statistic as implemented in ArcGIS 10.1. This method tests the null hypothesis that spatial association between neighboring high ('hot spots') or low ('cold spots') values is due to random clustering and is given as:

$$G_i^* = \frac{\sum_{j=1}^{n} w_{i,j} x_j - \bar{X} \sum_{j=1}^{n} w_{i,j}}{S \sqrt{\frac{\left[n \sum_{j=1}^{n} w_{i,j}^2 - \left(\sum_{j=1}^{n} w_{i,j}\right)^2\right]}{n-1}}} \tag{2}$$

where x_j is the attribute value for feature $j, w_{i,j}$ is the spatial weight between the features i and j, n is equal to the total number of features and

$$\bar{X} = \frac{\sum_{j=1}^{n} x_j}{n} \tag{3}$$

and

$$S = \sqrt{\frac{\sum_{j=1}^{n} x_j^2}{n} - (\bar{X})^2} \,. \tag{4}$$

Based on a nearest-neighbor approach, local patterns of fishing effort and catch patterns are identified and compared to what is generally observed across the whole study area.

Neighborhoods were defined by polygon contiguity (common boundary). As a result, the Getis-Ord *Gi** statistic returns Z-scores (also known as standard scores), which give information about whether the observed clustering between neighboring points can be attributed to random spatial processes, given their distance and value relative to the mean. A high positive *Z*-score (> + 1.65) represents statistically significant spatial clustering of high values (hot spot), a low negative *Z*-score (< - 1.65) represents statistically significant spatial clustering of low values (cold spot). A *Z*-score near zero indicates no apparent spatial clustering.

To explore changes in spatial clustering over time, the data was bisected into two 8-year intervals: 1990-2002 and 2003-2015. The year 2002 represents an even split of the time series and coincides with the time of improved reserve enforcement that ended widespread illegal fishing (Kliffen & Berkes 2015) several years after designation of the Galápagos Marine Reserve. Using the nominal date of designation (1998) as a split point returns similar results. Long-term temporal trends in fishing effort, catch, and CPUE were approximated using local polynomial regression. The annual total across all cells in the study area was calculated, and log-transformed to account for overdispersion. To test for significant differences between time trends in the hot spot and the study area, repeated-measures ANOVAs were performed. Separate repeated-measures ANOVAs were run for each of our variables of interest (fishing effort, catch, and CPUE), which were included as dependent variables. Location (grid cell) was included in each model as a single repeated measures factor with two levels (hot spot and study area).

Data for temporal trends of relative annual recruitment and spawning biomass ratio of yellowfin tuna were extracted from Minte-Vera et al. (2014) using WebPlotDigitizer version 3.12. Yellowfin tuna spawning biomass ratio as well as CPUE data were not normally distributed (Shapiro-Wilk normality test p = 0.004 and p = 0.002 respectively). Hence a non-parametric Spearman's rank correlation test was conducted using R Version 1.0.136 to contrast annual CPUE and spawning biomass ratio.

FAD data

The data on the use of FADs was split into two time series, 1990 – 2002 and 2003 – 2015, using 2002 as the split point as explained above for the IATTC observer data. Sets were summarized by set type (dolphin-associated, not associated, floating object [FAD] associated) and location (grid cell).

AIS data

Fishing set locations were extracted from the AIS data using the purse seine algorithm for satellite AIS data developed by de Souza et al. (2016). Briefly, this algorithm identifies fishing events based on low vessel speed (< 2.5 knots) and operational time (daylight only), with an accuracy of 97% against expert-labelled data (de Souza et al. 2016). Strings of continuous fishing activity were grouped into bursts and identified as individual sets based on the average set length as denoted by Walker & Bez (2010) with a minimum set time of 10 minutes.

To analyze the distribution of sets in relation to the reserve, set locations and their distance to the boundary were calculated and binned into areas of increasing distance from 0-10 km, 10-20 km, 20-50 km, 50-100 km, 100-200 km and 200-400 km. A nearest-neighbor analysis was conducted to test for spatial autocorrelation using the Nearest Neighbor Index in QGIS 2.18. To compare fishing effort as seen from IATTC and AIS data, a hot spot analysis using AIS data was based on the hot spot for fishing effort as determined from the IATTC observer data from 2003-2015 including 21 cells with a significantly higher positive Z-score as described above and shown in Fig. 3.1 (hatched area).

Results

IATTC observer data

Statistically significant clusters of 1x1° cells of high catch, fishing effort, and catch-per-unit-effort (CPUE) were identified for the time period prior to reserve enforcement (1990-2002) to the northwest, northeast, and (for catch and effort only) south of the Galápagos Marine Reserve (Fig. 3.2, Z-scores >1.65). For the period after Galápagos Marine Reserve designation and enforcement (2003-2015), catch, fishing effort, and CPUE all clustered around a large, statistically significant fishery 'hot spot' directly adjacent to the western and southwestern Galápagos Marine Reserve boundary (Fig. 3.2, Z -scores >1.65). This hot spot is located downstream of the reserve given the prevailing east-west currents. Cold spots of lower than expected catch, fishing effort, and CPUE were more consistently located to the south and southeast of the Galápagos archipelago (Fig. 3.2, Z -scores < -1.65).

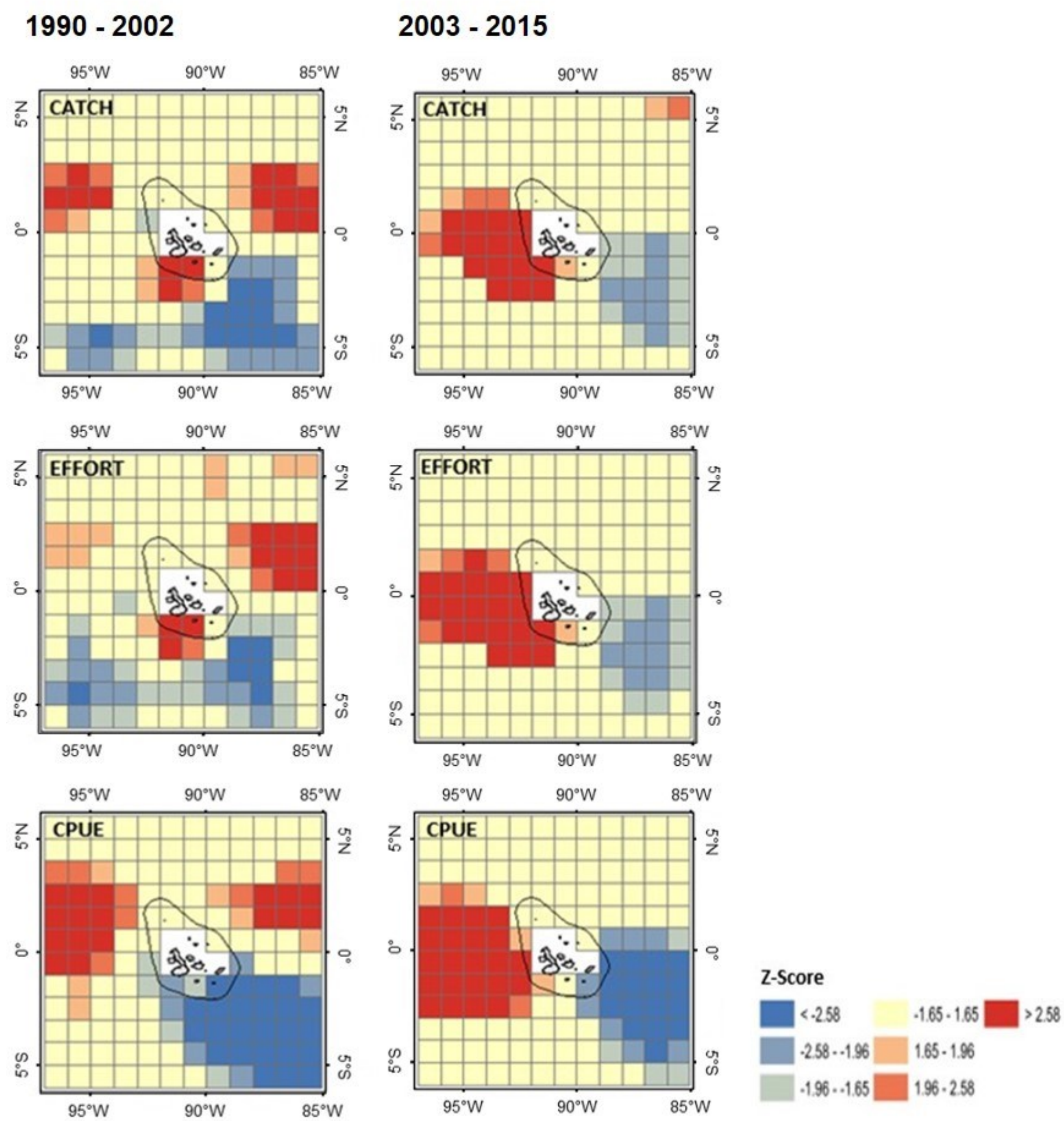

Fig. 3.2 Spatial trends in IATTC tuna fishing observer data. Clustering of purse seine catch, fishing effort, and CPUE were identified around the Galápagos Marine Reserve for the years 1990 – 2002 (left panels) and 2003 - 2015 (right panels). Z-scores less than -1.65 denote statistically significant cold spots (blue). Z-scores >1.65 indicate statistically significant hot spots (red). Galápagos Marine Reserve boundary is shown as a black outline around the Galápagos Islands. Note the concentration of effort, catch and CPUE hot spots downstream of the Galápagos Marine Reserve 2002-2015. Raw data are displayed in Fig. S1**.** Map data from Natural Earth and MarineRegions.org, and IUCN and UNEP-WCMC, The World Database on Protected Areas (WDPA), May 2016.

In addition to the spatial clustering of catch, fishing effort, and CPUE we observed progressive divergence of temporal trends in tuna fishing that coincided with Galápagos Marine Reserve enforcement (Fig. 3.3). Catch and fishing effort increased throughout the study period in the study area (Fig. 3.3 A: r^2 = 0.75, 3B: r^2 = 0.77) and CPUE showed a less consistent increase over the same time period (Fig. 3.3 C: r^2 = 0.65), with an initial increase followed by a decrease. This CPUE pattern is mirrored by recruitment and biomass data from regional IATTC stock assessments, with strong correlation between observed CPUE and assessed spawning stock biomass (Spearman's rank correlation, p=0.01484, Fig. A.2). Overall trends in recruitment, biomass, and CPUE are similar across principal tuna species and thought to reflect large-scale oceanographic changes that drive variations in stock productivity (Aires-da-Silva & Maunder 2014, Minte-Vera et al. 2014).

Comparing time trends of catch, fishing effort, and CPUE in the area minus the hot spot and the hot spot itself, both catch and fishing effort showed a significantly larger increase within the hot spot over time (repeated-measures ANOVA: catch: F = 10.55, p = 0.0033, n = 26; fishing effort: F = 7.46, p = 0.0114, n = 26; CPUE: F = 9.415, p = 0.0052, n = 26). With respect to the timing of Galápagos Marine Reserve enforcement across the whole study area we observed an average increase of 52% in catch and 75% increase of fishing effort per 1x1° grid cell respectively, after designation and enforcement of the Galápagos Marine Reserve (2003-2015) relative to 1990 – 2002. Over the same time frame CPUE decreased by less than 1%. Further, in the hot spot we observed a greater than 200% increase of both average catch and fishing effort per 1x1° grid cell, as well as a 7% increase of CPUE in the time period after Galápagos Marine Reserve enforcement relative to the preceding time period. In contrast, not taking the hot spot into account, CPUE decreased by 10% (catch increased by 2%, fishing effort by 25%). Fishing effort, catch and CPUE from IATTC observer data aggregated at 1x1° was not significantly correlated spatially (Moran's I test: sd = -0.97, p = 0.8341).

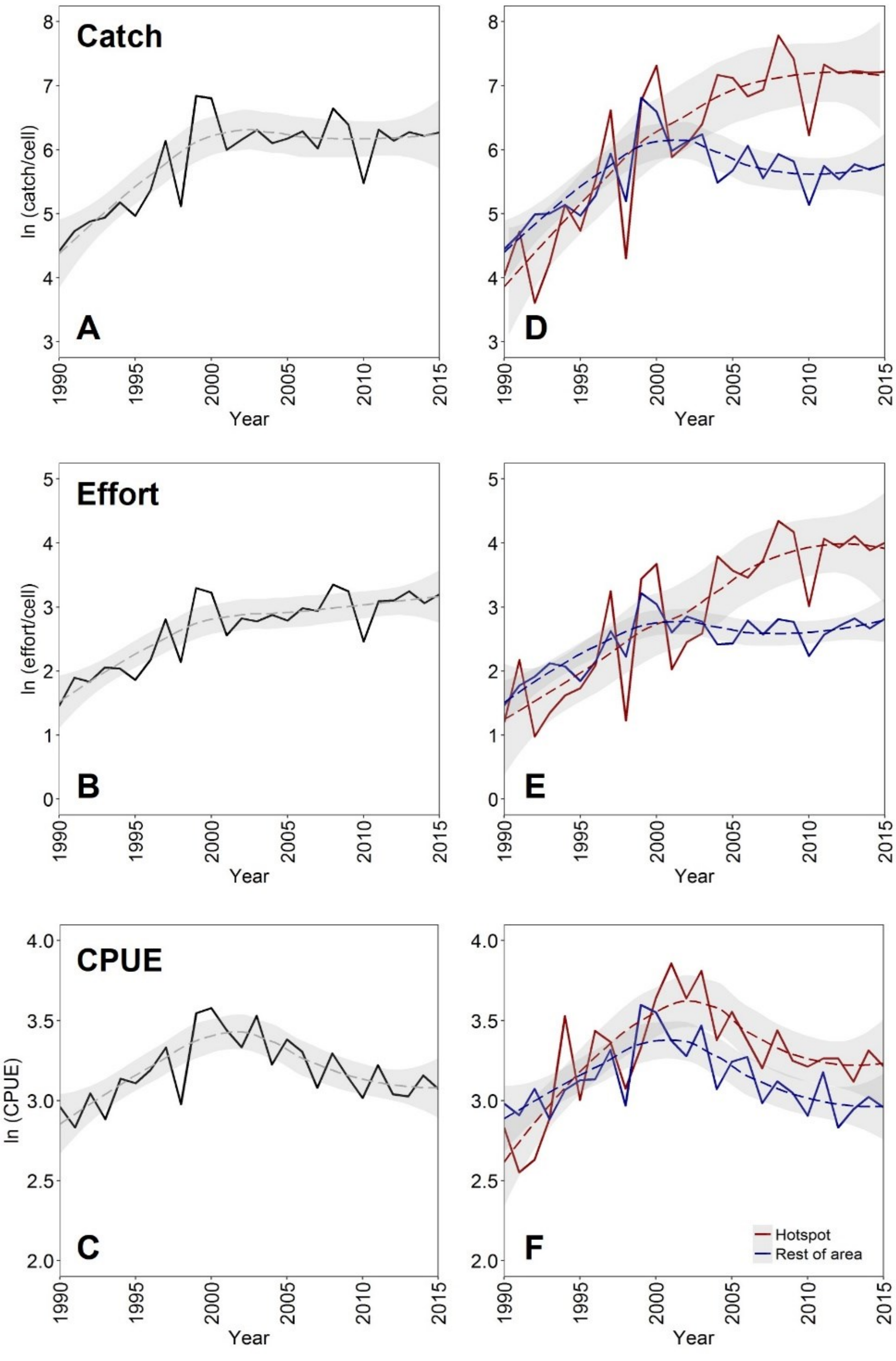

Fig. 3.3 Temporal trends in IATTC purse seine observer data. Shown is the natural logarithm of tuna catch (**A, D**), fishing effort (**B, E**) and CPUE (**C, F**) for tuna purse-seine fisheries from 1990–2015 across the entire study area (black lines in **A-C**) or differentiated between the fisheries hot spot (red lines) and the remainder of the study area (blue lines in **D-F**). Dashed lines and shaded area indicate polynomial regression model fit and 95% confidence interval, respectively. Note that trend lines diverge progressively after the marine reserve was enforced in 2002.

AIS data

Between January 2011 and October 2015, a total of 3,391 AIS-carrying vessels passed through the study area, 66 of which were identified as purse seiners. This is about half the number of vessels carrying an IATTC observer and 24% of all purse seiners currently active and registered with IATTC for the Eastern Tropical Pacific (272 purse seine vessels in 2017). The vessels were flagged to 10 different nations with Ecuadorian vessels representing the majority (27%). Fishing activity was detected for 49 of these vessels with a total of 664 fishing events or ‘sets’ (Fig. 3.1 B). Vessel size ranged from 416 mt up to 2,799 mt. All vessels observed in the area were registered with IATTC.

The spatial patterns of purse seine sets extracted from the AIS data (Fig. 3.1) were non-randomly distributed (Nearest Neighbor Analysis, Z = -18.24). Many sets were closely associated with the reserve boundary: density within the first 20 km around the reserve was at least four times larger than the average across the study area (Fig. 3.4). Across the entire fishing hot spot displayed in Fig. 3.2 (21 cells of varying distance to the reserve boundary) 170 purse seine fishing sets were detected using AIS, which represents twice the density of sets (0.0006 sets km^{-2} or 8.1 sets $cell^{-1}$) in the hot spot, as compared with the rest of the study area (0.0003 sets km^{-2} or 4.02 sets $cell^{-1}$) (Fig. 3.4).

Fishing effort per month varied strongly over the study period in both data sets with very few sets obtained from the AIS data during the years 2011 and 2012 and some months without any observed sets at all (Fig. A.4) likely due at least in part to poor satellite coverage at that time. The trends of fishing effort are similar in both data sets from 2012 onwards with peak fishing periods from January to July 2013 and June 2014 to January 2015.

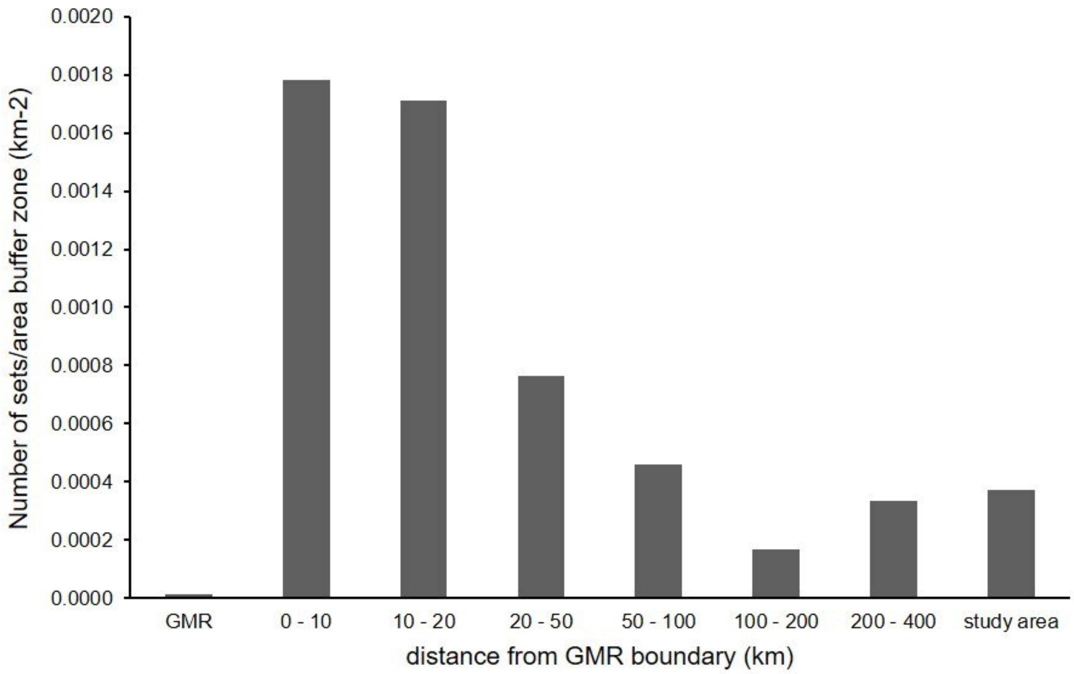

Fig. 3.4 Density of purse seine fishing sets detected by AIS tracking data 2011-2015, binned by distance from the marine reserve boundary. Note the remarkable aggregation of fishing effort within 20 km of the reserve boundary. Sets detected inside the Galápagos Marine Reserve (GMR) were also located very close to the boundary. Average density of sets for the total study area is displayed for comparison.

FAD data

Between 1990 and 2015 a total of 95,273 sets on tuna were recorded by IATTC within the study area with a total catch of 1,907,034 tons. About 36% of these sets were on floating objects or FADs (artificial and naturally occurring, Table A.1). The use of FADs in the study area has increased over the study period. In the time period from 1990 to 2002, about 10% of all sets were on FADs whereas from 2003 to 2015 26% were set on FADs. In contrast to the rest of the study area where the majority of sets have been on FADs between 2002 and 2015, most sets in the hot spot area (compare Fig. 3.1) are set on free schools (Fig. 3.5). Thus, while reliance on FADs has increased overall, this is seen to a much lesser extent in the areas with the most fishing effort and catch, i.e. the hot spot area adjacent to the marine reserve, indicating greater availability of free tuna schools.

Overall CPUE (tons per set) was higher on floating objects (25 tons/set) than on free schools or schools associated with dolphins (20 tons/set, Table A.1 and Fig. A.5). Catch per unit effort has decreased for all sets and for sets only on floating objects between the two periods from 1990 to 2002 (21 tons/set all sets and 32 tons/set floating objects only) and 2003 to 2015 (19.5 tons/set all sets and 23 tons/set floating objects only, Table A.1). Although we cannot rule out that some FADs were set in such a way that they would drift through the reserve, collecting tuna along the way, no data or reports exist that indicate such a practice.

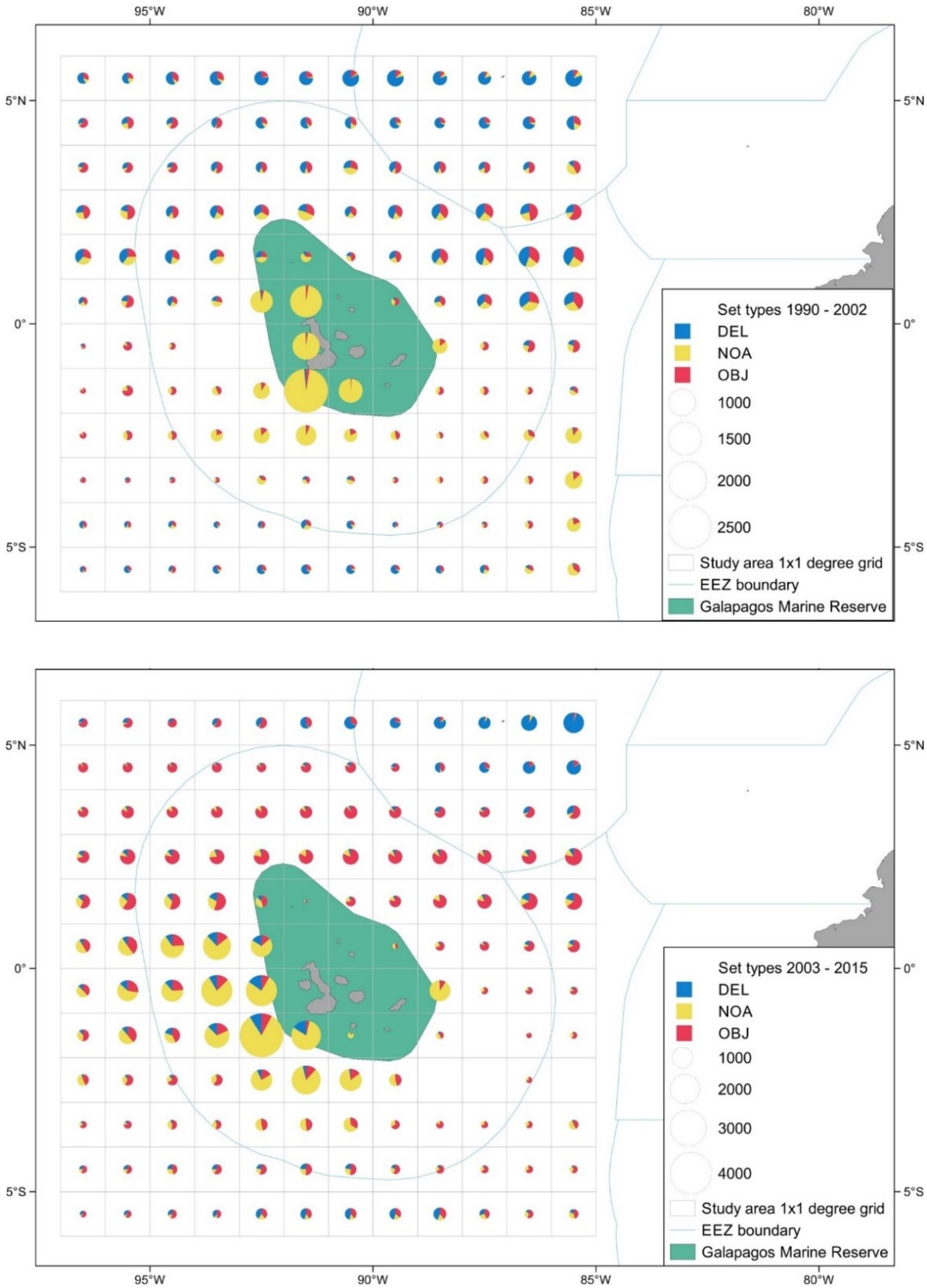

Fig. 3.5 Distribution of tuna purse seine set types. Types of tuna sets within the study area (grey grid) around the Galápagos Islands and the Galápagos Marine Reserve (green) before full enforcement of the reserve 1990 – 2002 (**A**) and after 2003 – 2015 (**B**). Note the increased proportion of sets on floating objects (red) as well as the number of sets not associated to objects or dolphins (yellow) to the west and southwest of the Galápagos Marine Reserve. Exclusive Economic Zone boundaries shown in light blue. Map data from Natural Earth and MarineRegions.org, IUCN and UNEP-WCMC, The World Database on Protected Areas (WDPA), May 2016.

Discussion

Results of this study suggest that industrial tuna fisheries concentrate close to the Galápagos Marine reserve, and that this area has been supporting higher catches, effort and catch per unit of effort (CPUE) since reserve implementation and enforcement, but not before. While fisheries in the study area have increasingly relied on fish aggregating devices (FADs) to maintain viable catches, this practice has been much less commonly used in the fisheries hot spot close to the reserve, suggesting that larger, commercially viable free tuna schools still occur naturally in this area. The designation of the Galápagos Marine Reserve in 1998 and enforcement around 2002 initially met with strong, sometimes violent opposition (Denkinger & Vinueza 2014, Kliffen & Berkes 2015) and ended decades of commercial fishing in the highly productive waters around the islands (Schiller et al. 2014). Yet, as fisheries have appeared to thrive closer to the reserve, local industry support for the Galápagos Marine Reserve has grown more recently (Kliffen & Berkes 2015).

Our detailed analyses of vessel tracks, fishing locations and type of purse seine sets indicate a direct association of fishing effort with the reserve boundary, and progressively lower density of sets with increasing distance from the reserve (Fig. 3.4). In addition, overall fishing effort is increasingly concentrating around the reserve (Fig. 3.2) and in a regionally unique fisheries 'hot spot' west and southwest of the reserve, identified both in the IATTC observer and AIS data (Fig. 3.1 and Fig. A.3) as well as anecdotally by fishermen (Kliffen & Berkes 2015, Martínez-Ortiz et al. 2015). This hot spot receives nearly four times the density of fishing effort than the average for study area and sustains higher catches and CPUE despite an increasing concentration of fishing effort (Fig. 3.2). The area features deeper waters and complex currents, creating upwelling and frontal areas (Liu, Xie, et al. 2014) and represents favorable habitats for pelagic predators most of which range throughout the whole Eastern Pacific (Fiedler et al. 1991, Miller 2007, Worm & Tittensor 2011). It is now a preferred fishing ground for tuna purse seiners and longliners targeting mainly yellowfin and bigeye tuna as well as swordfish (*Xiphias gladius*) and a variety of sharks (Martínez-Ortiz et al. 2015). While effort increased in the hot spot, another fishing hot spot to the northeast of the reserve has disappeared in the early 2000s (Fig. 3.2). It can be assumed that some of the vessels previously fishing in that area are now fishing closer to the boundary of the reserve. Whether they have changed fishing location in response to the establishment of the reserve or whether they respond to a change in oceanographic conditions or fish stock distribution remains unclear.

Fishing sets in the hot spot close to the reserve boundary are largely done on free swimming schools of tuna, in contrast to surrounding areas where sets are commonly focused on floating objects such as artificial or naturally occurring FADs or schools associated with dolphins (Fig. 3.5). Free schools appear to be much more abundant along the reserve boundary and in the hot spot, reducing the need to rely on FADs. The majority of catch from sets on free schools has changed from yellowfin tuna to skipjack tuna over the last decade. Skipjack tuna is also most commonly caught in sets on floating objects, followed by bigeye tuna (Miyake et al. 2010) (Fig. A.5).

Tuna stock availability in the hot spot appears to be higher than in surrounding waters of the wider region, attracting higher fishing effort, supporting higher catches and dampening an overall declining CPUE trend (Fig. 3.3). Available stock assessments suggest that this trend might be connected to changing productivity regimes affecting recruitment and biomass trends, most notably for yellowfin tuna (Minte-Vera et al. 2014) (Fig. A.2). Following a strong recruitment peak around 1998 for all three main tuna species fished by purse seiners in the area (yellowfin, bigeye and skipjack tuna) stock assessments show declining recruitment and subsequent loss of spawning stock biomass especially for yellowfin and bigeye tuna around the year 2002, contributing to declining catches in the following years (Aires-da-Silva & Maunder 2014, Minte-Vera et al. 2014). Despite this overall productivity decline, after reserve implementation and subsequent enforcement in 2002, fishing effort and catches in the hot spot along the reserve boundary increased notably, in contrast to the rest of the study area (Fig. 3.3). While CPUE (and overall stock biomass) decreases gradually across the region after 2000 it remains higher in the hot spot after reserve enforcement. According to local ecological knowledge by fishermen, the area surrounding the Galápagos islands is believed to be a '*criadero*', a tuna spawning and breeding ground (Kliffen & Berkes 2015). Protection of habitats where vulnerable life stages aggregate, such as spawning grounds and nursery areas, is often assumed to produce disproportional reserve benefits (Halpern et al. 2004, Game et al. 2009) (Chapter 6). These factors might indicate a combination of favorable local environmental conditions and fisheries benefits that accrue downstream from a well-protected and well-placed reserve.

The use of previously unavailable high-resolution AIS data allowed us to investigate the fine-scale patterns of tuna fishing around the reserve revealing close attraction of fishing effort to the immediate reserve boundary (Fig. 3.4) receiving more than four times the density of sets than the area from 20 to 400 km from the boundary (Fig. 3.4, Fig. A.3). The hot spot lies downstream from predominant east-west currents, which may transport adult tuna as well as larvae and juveniles into the hot spot area (Reglero et al. 2014). Fisheries here potentially benefit from a spillover effect, explaining higher catches in the hot spot despite increasing effort (Fig. 3.3). 'Fishing the line' behavior has been observed around other spatial closures and is a potential indicator for spillover benefits in the form of more fish leaving the reserve that are available to fisheries nearby (Rijnsdorp et al. 1998, Johnson et al. 1999, Murawski et al. 2005, Kellner et al. 2007, Russ & Alcala 2011, Alemany et al. 2013, Van Der Lee et al. 2013). Spillover benefits in the area have been explicitly recognized by fishermen interviewed in a qualitative study by Kliffen and Berkes (2015). The declining CPUE trends overall, however, might indicate that any reserve spillover is not pronounced enough to reverse overall biomass trajectories for the region as indicated by relevant stock assessments (Aires-da-Silva & Maunder 2014, Minte-Vera et al. 2014).

The long-term trends discussed here are superimposed on large inter-annual fluctuations in fishing effort, catch and CPUE as seen in the IATTC data (Fig. A.1). These fluctuations are likely controlled by a number of factors, related to fleet behavior, seasonality (Sweet et al. 2007), and the strong variability associated with regional climate fluctuations, namely the El Niño-Southern Oscillation (ENSO) (Lehodey 2000) and Pacific Decadal Oscillation (Worm et al. 2005). In addition, fisheries management influences the distribution of purse seine vessels through periodic time-area closures for industrial purse seiners enforced since the early 2000s (Maunder & Harley 2006), potentially contributing to observed shifts in fishing effort (Fig. 3.2). Temporal closures are known to affect spatial as well as temporal dynamics of fishing effort and can lead to an intensification of fishing effort along closure boundaries and in closure areas especially after the seasonal opening of the closed zone (Murawski et al. 2005). For example, monthly set data from both IATTC and AIS show that purse seiners abide by a first fishing closure period August and September but remain partially active during a second closure period from November to January (Fig. A.4).

Through the combination of long-term, large-scale observer data and short-term, fine-scale AIS data we were able to obtain a reasonably comprehensive picture of purse seine fishing fleet behavior around the Galápagos Marine Reserve. Vessel monitoring using on-board observers has a long tradition and is ideal not only to obtain information about the activities of the fishing vessel but also detailed information on the catch. However, this method is limited by the numbers of observers available, the proprietary nature of the data, and the coarse scale at which data are reported. AIS data, especially satellite AIS, is an emerging tool in fishing vessel monitoring and surveillance (Cairns 2005, Carson-Jackson 2012, McCauley et al. 2016). While the system does not provide data on catches, vessels carrying an AIS transponder can be tracked on a fine scale in any part of the ocean. Data transmission is only limited by the number of vessels carrying AIS transponders and the number of satellites in an area receiving the AIS signals. Due to the differences in information transmitted with each dataset, the observer data and the AIS data are highly complementary and well-suited to be combined and contrasted as shown here. There have been increasing calls to make AIS use mandatory for all fishing vessels, and as such to create a more detailed and comprehensive picture of vessel activity that can be used by Regional Fisheries Management Organizations and other stakeholders (McCauley et al. 2016).

However, some important caveats and limitations remain: while observer coverage reaches 100% for IATTC capacity class 6 vessels in the last years (IATTC 2016), some smaller vessels do not carry an observer on board. Likewise, AIS coverage is now mandatory for all fishing vessels operating a motor within the Galápagos Marine Reserve, however, this regulation was only imposed in 2015. Small and artisanal vessels less than 10 tons and some industrial vessels less than 300 GT remain largely undocumented in both the IATTC and the AIS data sets. Furthermore, AIS vessel tracks can be incomplete due to insufficient satellite coverage and the AIS transponder can be manipulated, e.g. track locations or vessel identities can be manually altered. AIS data might therefore require some pre-processing.

Notwithstanding these inherent limitations, this is, to our knowledge, the first study examining interactions between a large MPA and an associated pelagic fishery. Many of the interactions seen in this study are likely shaped by the unique context and local factors such as oceanographic features and regional fisheries management, however, some patterns like the fishing the line have been seen for other, mostly smaller MPAs around the world (e.g. Murawski et al. 2005). While this work aims to explore an in-depth case study, a comprehensive analysis comparing interactions of fisheries with large MPAs more generally, would be desirable (Chapter 4).

Conclusion

In this study, the combination of on-board observer and satellite AIS tracking data provided a detailed picture of vessel behavior, fishing effort, and catches in the Eastern Tropical Pacific, and could be utilized more generally to monitor fisheries and conservation outcomes of large marine reserves in real time. We found that tuna purse seine vessels have reacted to the establishment of the Galápagos Marine Reserve and are increasingly fishing along the boundary of the reserve. Possibly due to a combination of reserve benefits on local stock availability and favorable habitat in this hot spot, tuna catches are higher than in surrounding areas since reserve enforcement, and fisheries there rely much less on the use of fish aggregating devices than in surrounding areas. These apparent benefits to fishers were realized despite an overall increase of fishing effort and declining tuna recruitment and productivity across the wider region. In aggregate, these results suggest that the Galápagos Marine Reserve has a net positive effect on associated pelagic fisheries and supports the case for establishing large-scale MPAs both as fisheries management and biodiversity conservation tools.

Acknowledgements

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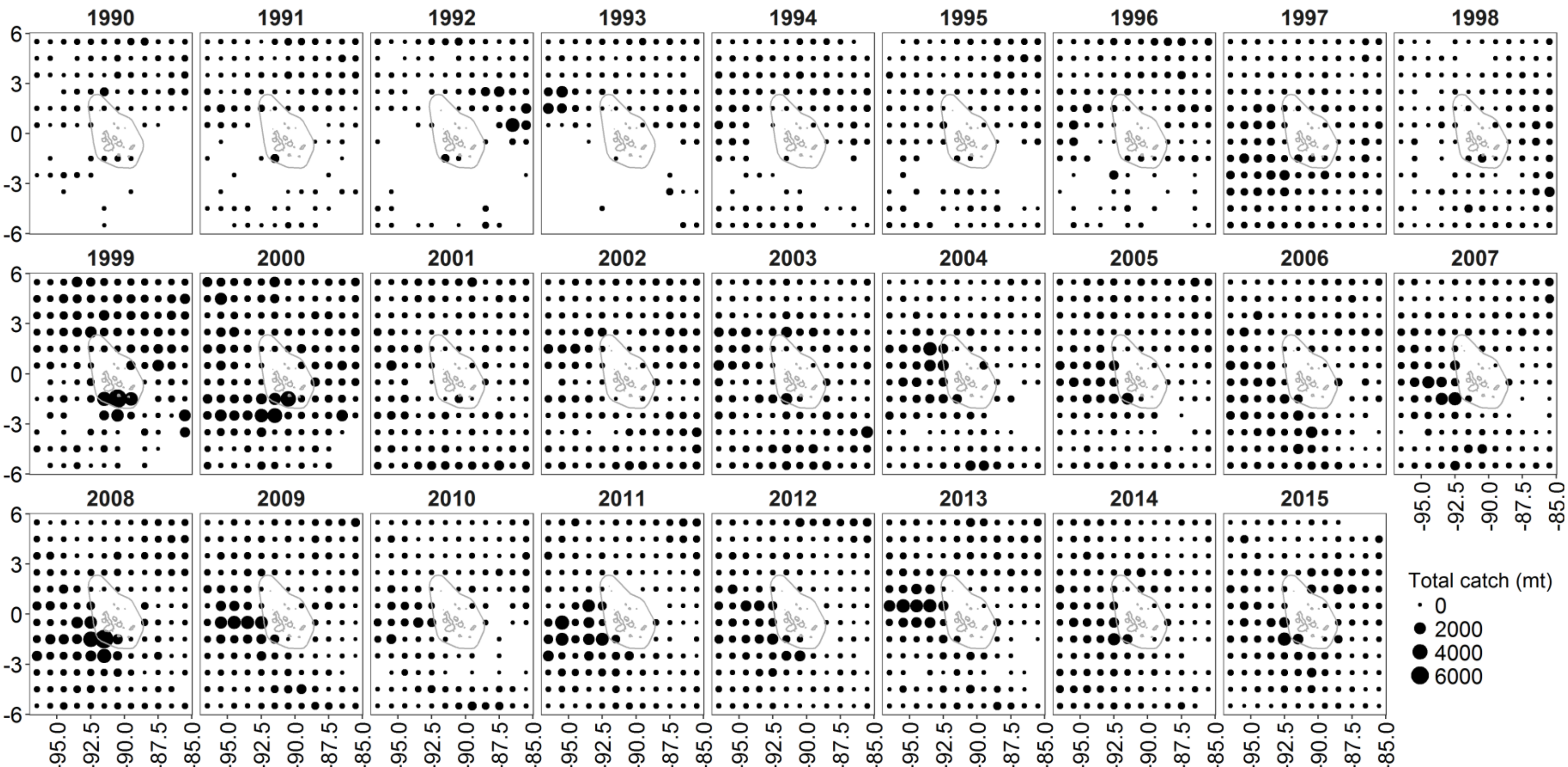
Author Contributions

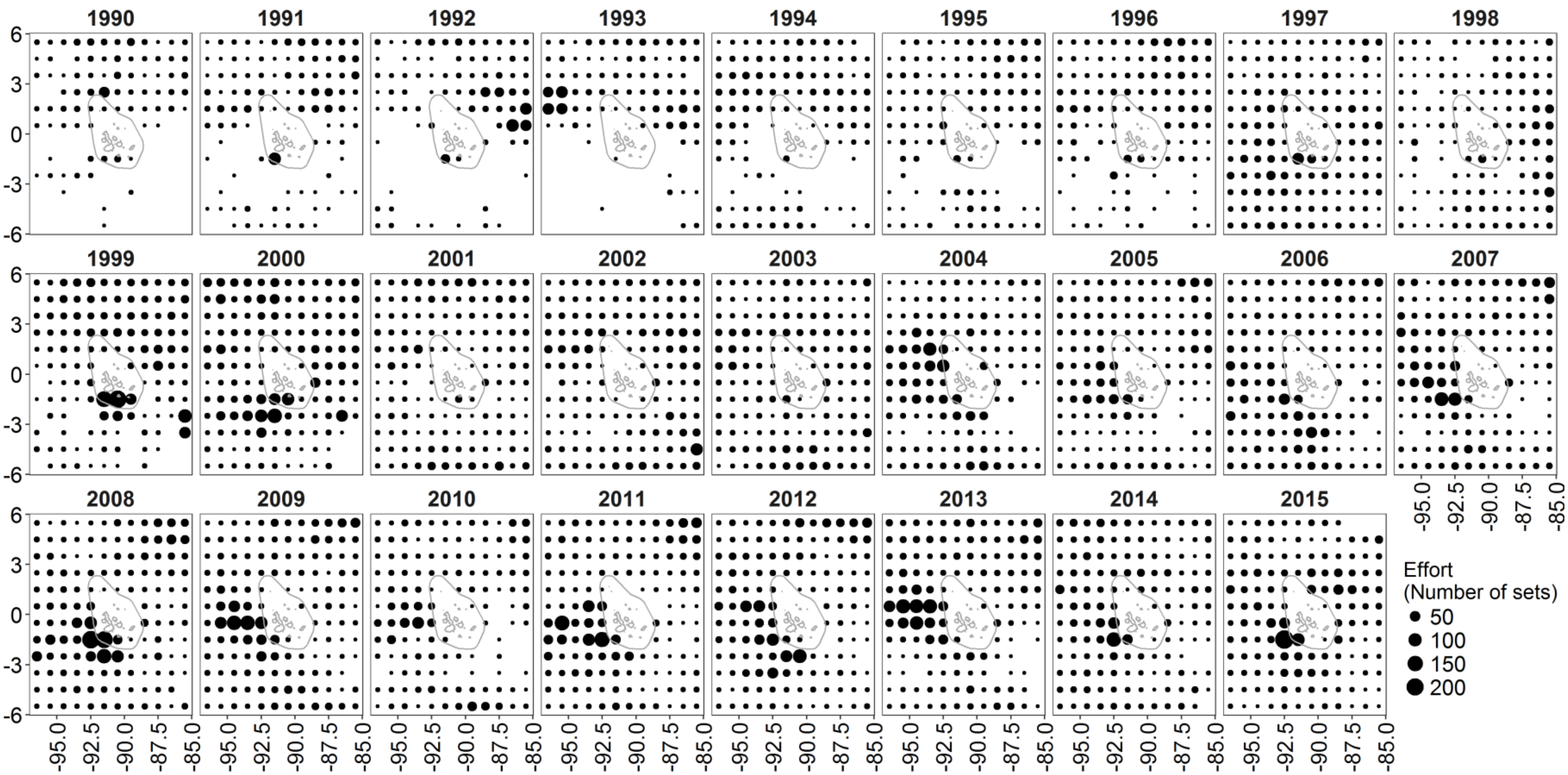
Concept and design by KB. Data preprocessing and analyses by KB and ABB, discussion and writing by KB, ABB and BW

Appendix A

a. Catch



b. Effort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2009
2010
2011
2012
2013
2014
2015
6
3
0
-3
-6
-95.0
-92.5
-90.0
-87.5
-85.0
Effort
(Number of sets)
50
100
150
200

c. Catch per unit effort

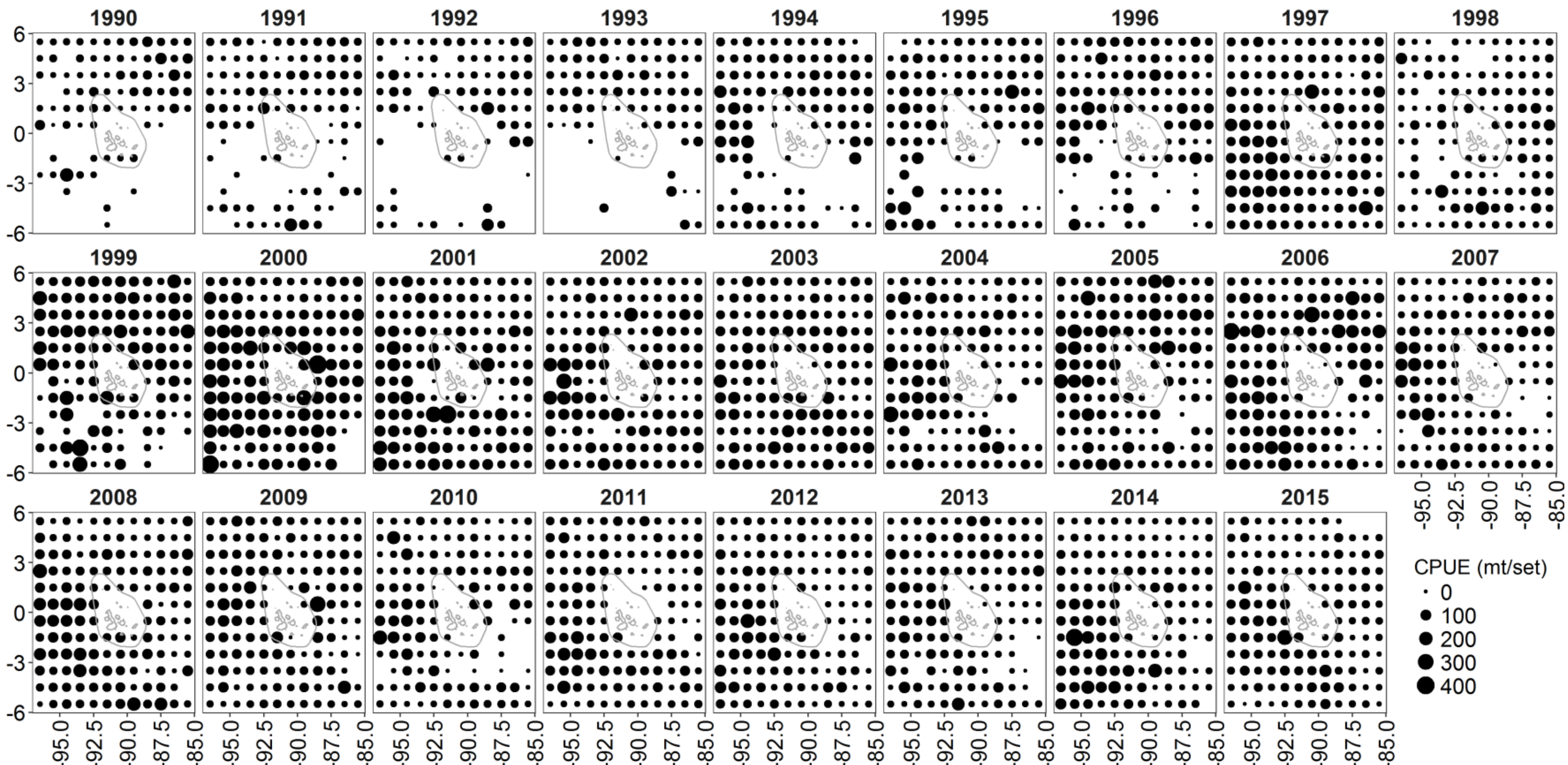


Fig. A.1 IATTC observer data by year. Shown are bubble plots of **a.** total catch, **b.** effort and **c.** catch per unit effort of observed purse seine vessels is shown combined for all tuna species in the study area from 1990 – 2015. Map data from Natural Earth and MarineRegions.org and IUCN and UNEP-WCMC, The World Database on Protected Areas (WDPA), May 2016.

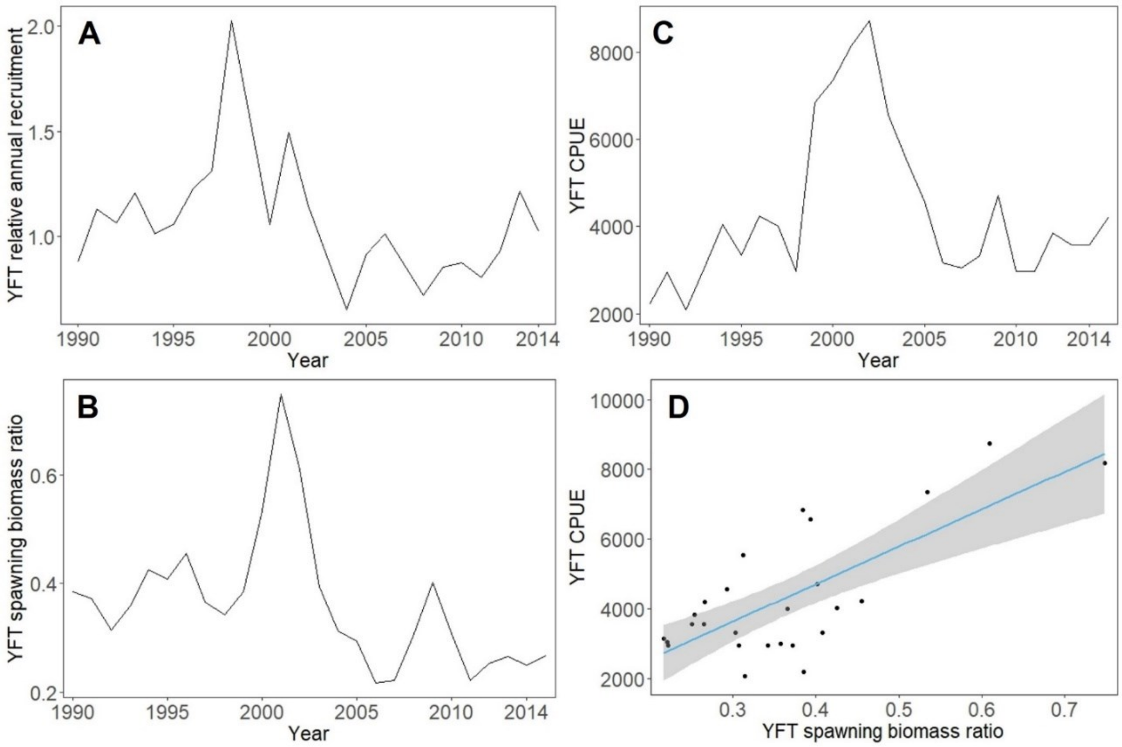


Fig. A.2 Temporal trends for yellowfin tuna stocks in the Eastern Tropical Pacific from 1990 - 2015. (**A**) Relative annual recruitment, (**B**) Spawning biomass ratio (relative to virgin population), (**C**) Catch-Per-Unit-Effort (CPUE) and (**D**) Correlation between spawning biomass ratio and CPUE. Blue line shows linear regression, grey area depicts 95% confidence interval. Spearman's correlation coefficient r_s = 0.48 and p = 0.015. Data for A and B taken from Minte-Vera et al. (2014).

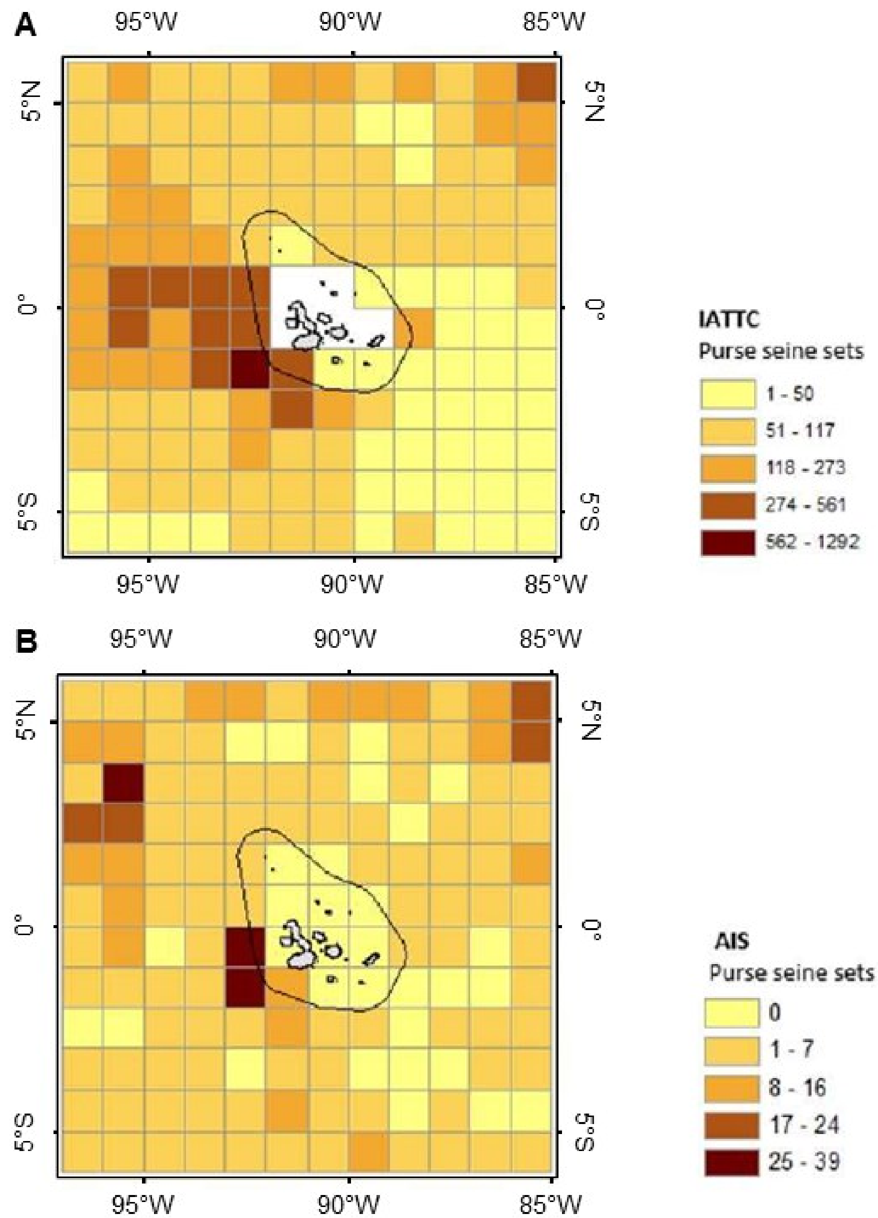


Fig. A.3 Spatial comparison of observer and AIS data. All purse seine sets from (**A**) IATTC and (**B**) AIS were aggregated to 1x1° cells within the study area between January 2011 and October 2015. Galápagos Marine Reserve boundaries depicted in black. Map data from Natural Earth and MarineRegions.org and IUCN and UNEP-WCMC, The World Database on Protected Areas (WDPA), May 2016.

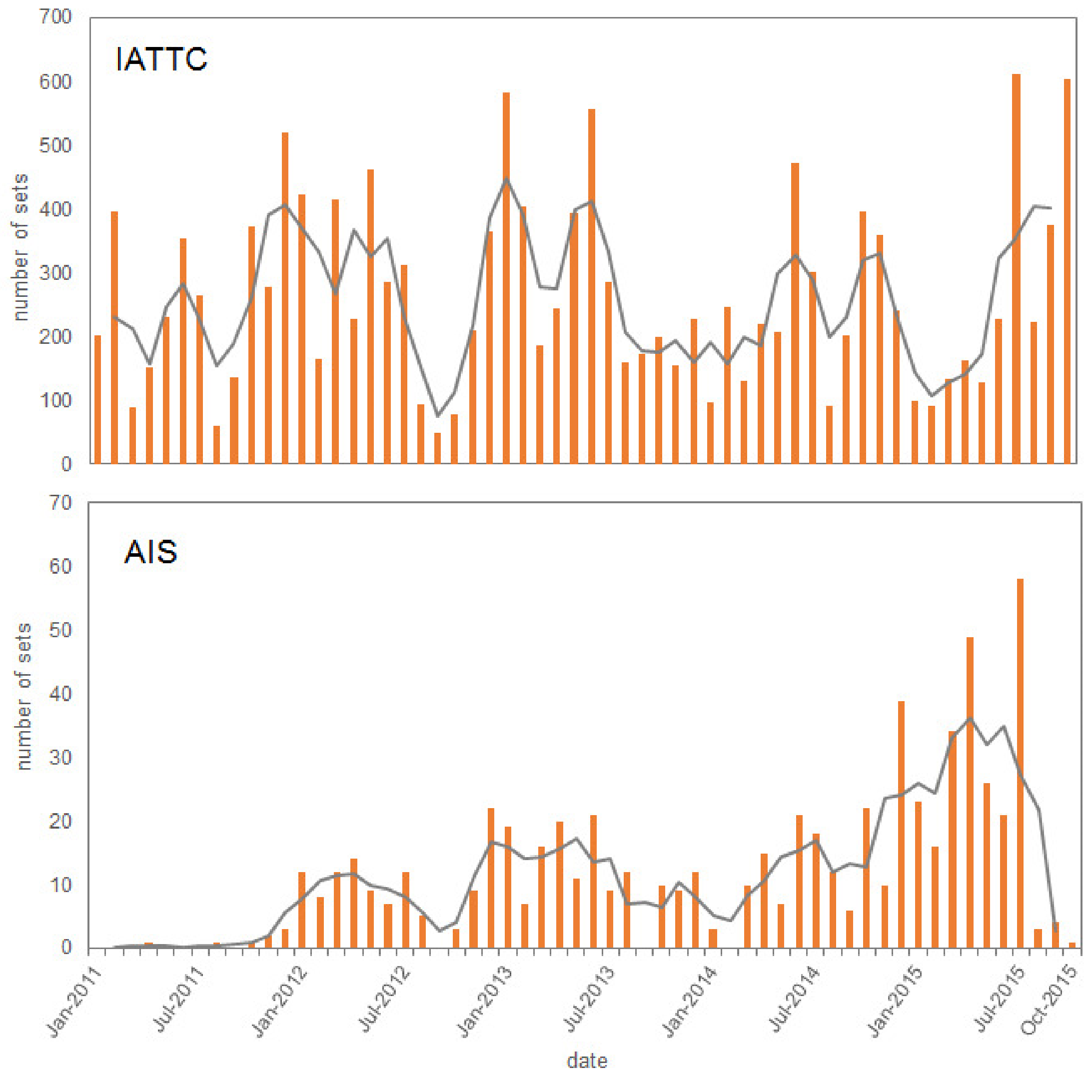


Fig. A.4 Temporal comparison of observer and AIS data. Shown are the number of purse seine sets from IATTC observer data and AIS data used in this study between January 2011 and October 2015. Orange bars: sets per month, grey line: 3 month moving average.

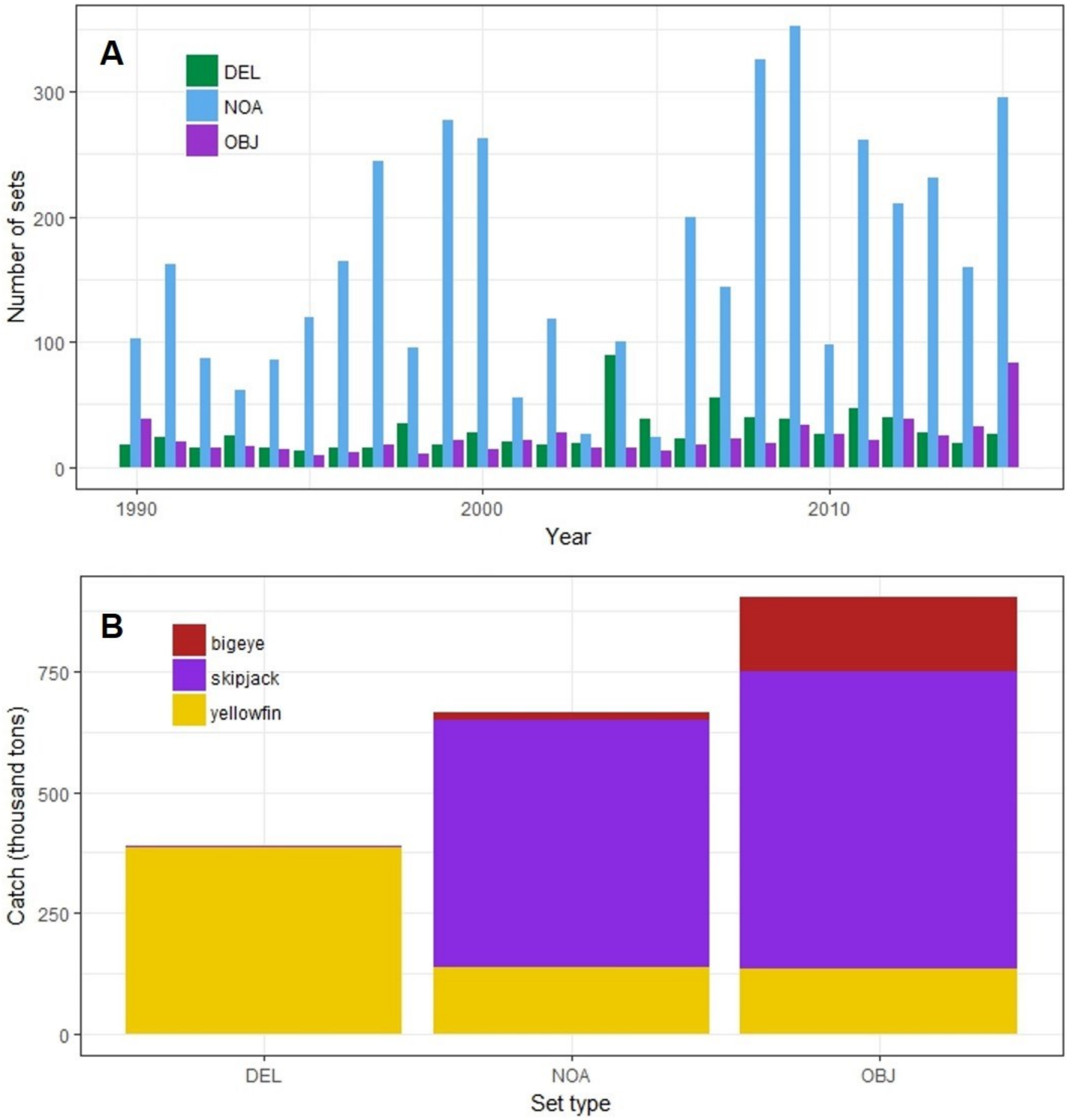


Fig. A.5 Purse seine sets in the study area from 1990 – 2015. (**A**) Number of sets per type over the time frame (DEL = dolphin associated, NOA = not associated/free school, OBJ = floating object associated), (**B**) Catch per species per set type (DEL = dolphin associated, NOA = not associated/free school, OBJ = floating object associated).

Table A.1 Set types within study area between 1990 and 2015.

	Sets	Catch (tons)	CPUE (tons/set)
Total number study area	95,273	1,907,034	20
Floating objects only	34,373	877,371	25
1990 – 2002 (all sets)	33,636	707,439	21
2003 – 2015 (all sets)	61,637	1,199,596	19.5
1990 – 2002 (floating objects only)	9,223	298,911	32
2003 – 2015 (floating objects only)	25,150	578,461	23

CHAPTER 4

INTERACTIONS OF LARGE-SCALE MARINE PROTECTED AREAS AND GLOBAL FISHERIES

Abstract

Spurred by international marine conservation targets, large-scale marine protected areas (LSMPAs) are increasingly being established around the globe to counter marine biodiversity declines, promote ecosystem resilience, and source benefits to global fisheries and coastal communities. Currently, established and promised LSMPAs make up 95% of the global marine protected areas (MPAs) aimed at biodiversity protection. Although fishing is restricted or forbidden across these large areas, little is known how LSMPAs and surrounding fisheries interact. Here, we analyse fishing effort around thirteen LSMPAs (>100,000km^2 in size and established on or before 2015) using Automatic Identification System (AIS) vessel tracking data from 2015 – 2017. Applying random forest models on more than 166,000 data points and over 5.8 million fishing hours, we examine how management status, as well as environmental, physical and other characteristics influence patterns of fishing effort around LSMPAs. We find that the behaviour of fishing fleets around LSMPAs is highly variable, with fishing intensity varying by a factor of 10. Observed fishing effort inside LSMPAs was generally low (median 0.01 hours fished/km^2), eight times less than in nearby fished areas (median 0.08 hours fished/km^2). Fully protected LSMPAs had twelve times less fishing effort inside than partially protected ones. The primary driver of overall fishing effort patterns appears to be maritime borders and jurisdictions, specifically the distance the from the MPA boundary to the high seas. Fishing effort appears to be primarily aggregated around the boundaries of older MPAs (15+ years). In aggregate, these results show that the spatial distribution of fishing effort around LSMPAs is influenced by two equally important factors: 1) long-established maritime zoning between territorial waters and the high seas and 2) the age of MPAs. This new insight about fishing behavior derived from AIS-tracking can hopefully help to guide management

of current MPAs and the planning of future MPAs as well as inform large-scale conservation efforts in areas beyond national jurisdiction.

Introduction

The current global trend to establish large-scale marine protected areas (LSMPAs, >100,000 km^2) (UNEP-WCMC & IUCN 2016) is accompanied by constructive public and scientific debate about best strategies for leveraging the power of LSMPAs to promote the protection of biodiversity, sustain fisheries, and meet the goals of drawing wealth and food from our global oceans (O'Leary et al. 2018). LSMPAs have become a growing part of the seascape and a diverse marine conservation toolbox: By 2018, 39 LSMPAs had been either designated (n=32) or promised (n=7), ranging from 109,000 to over 5,000,000 km^2 in size (Fig. 4.1). All but four LSMPAs were created after 2008 (Fig. 4.1 A). Taken together, all 39 LSMPAs established or promised by 2018 cover more than 25 million km^2, or about 7% of global ocean area (Fig. 4.1 C). For comparison all other 15,285 MPAs globally combined account for roughly 1 .3 million km^2, or about 0.3% of ocean area in 2018. This means that if promised areas get established LSMPAs will encompass up to 95% of all ocean are included in MPAs. However, their fairly recent nature (Table 4.1) necessitates a re-evaluation of ecological and socio-economic lessons learnt from a rich body of previous analyses mostly focused on smaller, coastal marine protected areas (MPAs) (Roberts et al. 2001, Russ & Alcala 2011, Costello 2014, Di Franco et al. 2016, Di Lorenzo et al. 2016, Giakoumi et al. 2017). Large-scale MPAs differ in many fundamental ways from previously established smaller MPAs: they are often located in remote areas, cover a range of habitats from shallow reefs to deep sea ocean floor, and due to their size encompass multiple ecosystems. It was found that existing MPAs of a size of 30,000 km^2 and more harbor a great proportion of biodiversity as assessed by Davies et al. (2017) and furthermore are a potential tool to meet future challenges such as extending fishing effort to meet an increasing demand for seafood (FAO 2018), as well as a changing climate (Davies et al. 2017, 2018, O'Leary et al. 2018). LSMPAs offer a variety of advantages compared to smaller

MPAs: due to their size, they protect a variety of biologically connected ecosystems within their area and are more likely to cover a significant fraction of species distribution ranges, including potential shifting ranges due to changing climate (Roberts et al. 2017, O'Leary et al. 2018). Another, often more implied benefit of LSMPAs is the creation of benefits for fisheries through enhancement of fish stocks and subsequent 'spillover' of adult fish and larvae from LSMPAs, a process that could be benefitting surrounding fisheries (Di Lorenzo et al. 2016). While spillover benefits are well know in smaller MPAs (e.g. Kellner et al. 2007, Russ & Alcala 2011, Van Der Lee et al. 2013), the novelty of LSMPAs creates uncertainty about their potential costs and benefits (Ban et al. 2017, Boerder et al. 2017, Gill et al. 2017, White et al. 2017, Bucaram et al. 2018). In the light of systematic conservation planning and in order to achieve meaningful protection of global biodiversity, the value of placing LSMPAs in remote areas that are often experiencing less fishing intensity has been questioned (Craigie et al. 2014, Devillers et al. 2014). Quantitative knowledge about potential benefits and drawbacks would thus constructively inform conversation about the impacts of LSMPAs on various stakeholders such as fisheries (Costello 2014, Di Lorenzo et al. 2016).

Previously, it has been challenging to study the effects of spatial protection over vast, remote areas. However, the advent of novel vessel tracking and surveillance technologies such as the Automatic Identification System (AIS) have enabled new ways to monitor and study the distribution of fishing effort on a global scale (Natale et al. 2015, de Souza et al. 2016, McCauley et al. 2016, Dunn et al. 2018, Kroodsma et al. 2018). This new data can be used to create hypotheses about the interactions of LSMPAs and fisheries by tracking the disproportionate presence of fishing activity on the perimeter of LSMPAs (Stelzenmüller et al. 2008, Boerder et al. 2017, Bucaram et al. 2018). Equally importantly, AIS data can provide useful and novel insight into how the geographic, political, and environmental context of LSMPAs shapes the distribution patterns of fishing effort. For example: Location (e.g. distance to coast or next port), social and political restrictions such as maritime zoning restricting access (Exclusive Economic Zones vs high seas, MPAs and fisheries closures), oceanographic (e.g. depth), economic (e.g. fuel price) and environmental (e.g. primary productivity) (Stelzenmüller et al. 2008, Edgar et al. 2014, Gill et al. 2017). To maximise conservation potential of LSMPAs and minimize negative impacts on stakeholders, it is vital to understand the role these factors play to design, manage, and enforce LSMPAs accordingly. This has been done for a range of smaller MPAs in various regions (e.g. Claudet et al. 2008, Edgar et al. 2014, Friedlander et al. 2017, Gill et al. 2017), however, has not been determined solely considering the world's largest MPAs, their unique setting, and management challenges.

Here, we examine the distribution of fishing effort around thirteen LSMPAs across the globe using a three-year dataset of worldwide fishing effort derived from AIS data. We analyze three aspects of spatial interactions between LSMPAs and fisheries: i) fishing inside LSMPAs, ii) fishing effort associating with MPA boundaries, and iii) the general patterns of fishing effort around LSMPAs. We ask (1) how fishing effort around LSMPAs behaves over space and time, (2) how MPA features influence fishing effort in a 500 km radius around its boundary, and (3) whether fishing effort concentrates close to LSMPA boundaries due to possible spillover effects?

Materials and Methods

Study areas

We selected thirteen MPAs larger than 100.000 km^2 for this study (Table 4.1). As we are analyzing AIS data starting in 2015, we excluded very recent areas established after 2015, those lacking information required in this study such as management, and those with little or no fishing effort in their vicinity (outside of 95% confidence intervals; <0.031 fishing hours/km^2, see Table 4.1 B) due to a shortage of data in these areas.

Table 4.1 Overview of large-scale marine protected areas analyzed. Surface area is based on boundaries of LSMPAs prior to 2015.

Original year designated	MPA	Designating country	Location	Surface area (km2)
1975	Great Barrier Reef Marine Park	Australia	Southwest Pacific	344,000
1998	Galápagos Islands Marine Reserve	Ecuador	Eastern Tropical Pacific	178,000
1999	Macquarie Island Marine Park	Australia	Subarctic Pacific	162,000
2000	Papahānaumokuākea Marine National Monument	US	North Pacific	1,508,859
2008	Phoenix Islands Protected Area	Kiribati	Central Pacific	408,250
2009	Marianas Trench Marine National Monument	US	Northwest Pacific	246,608
2009	Pacific Remote Islands Marine National Monument	US	North and Central Pacific	1,269,094
2010	Chagos Marine Protected Area	UK	Central Indian Ocean	639,661
2012	Coral Sea Marine Park	Australia	Southwest Pacific	989,836
2012	Norfolk Marine Park	Australia	Southwest Pacific	188,444
2012	Lord Howe Marine Park	Australia	Southwest Pacific	110,126
2012	Argo-Rowley Terrace Marine Park	Australia	East Indian Ocean	146,003
2014	Natural Park of the Coral Sea	France	Southwest Pacific	1,291,000

Shapefiles for analysis in ArcGIS (version 10.5) were extracted from the World Database on Protected Areas (UNEP-WCMC and IUCN (2017) Marine Protected Planet). For cases where MPA area or zoning have changed over time, the MPA shape in 2015 was used. All area calculations are based on a Winkel Tripel projection, a modified azimuthal global map projection.

The study areas extend 500 km outwards from each MPA boundary, an area roughly corresponding to the spatial scale over which dispersal and recruitment patterns correlate for many marine species (Myers et al. 1997). To examine potential attraction of fishing effort to MPA boundaries this area was further subdivided into seven zones extending outwards from the MPA boundary from 0 – 20 km, 20 – 40 km, 40 – 80 km, 80 – 160 km, 160 – 320 km and 320 – 500 km as well as a zone comprising the area within the MPA. For more fine-grained analyses zones ranging from 0 – 2 km, 2 – 4 km, 4 – 6 km, 6 – 8 km, 8 – 10 km, 10 – 12 km, 12 – 14 km and 14 – 16 km from LSMPA boundaries were also used.

Characteristics of each MPA were compiled from available data such as MPA management plans, public databases (Protected Planet, MPAtlas, IMO Global Integrated Shipping Information System [GISIS], CIA World Factbook, World Bank Open Data, United Nations Development Programme Human Development Reports) as well as scientific literature (e.g. Gill et al. 2017, O'Leary et al. 2018), RFMO publications, and expert opinion (overview in Table B.1) and are summarised in Table B.2.

AIS data

Data on fishing activity is based on observations from the Automatic Identification System (AIS). AIS was designed as a collision avoidance system and consists of a transponder-receiver connected to a vessel's Global Positioning System (GPS), sending ship information such as location, speed, and course over ground to surrounding ships carrying AIS devices as well as to ground stations or satellites. The AIS positions recorded through these stations can be used to track vessel routes and analyse their activity (Natale et al. 2015, de Souza et al. 2016, James et al. 2018, Kroodsma et al. 2018).

AIS data was extracted in a 0.1 by 0.1 degree grid format from Global Fishing Watch (GFW) using results from the neural net version 7. By means of a set of algorithms as well as neural network techniques, GFW identifies ship types based on a comparison to vessel registries as well as ship behaviour. Fishing vessels are further classified by fishing gear type and their fishing activity is detected using machine learning neural network analyses as described by Kroodsma et al. 2018. A grid of 0.1 by 0.1 degree cells was chosen to best represent the spatial footprint of fishing effort in as much detail as possible, avoid spatial smoothing of the data due to coarse resolution (Dormann et al. 2007), and comply with computational restrictions. For five selected case studies showing potential spillover effects additional data was extracted on a 0.01 by 0.01 degree grid to enable more detailed analysis.

Environmental and physical parameters

For each grid cell where fishing was detected corresponding depth (http://www.naturalearthdata.com/downloads/10m-physical-vectors/), sea surface temperature (NASA Ocean Biology [OB.DAAC, 2014]. Mean annual sea surface temperature for the period 2009-2013), ocean productivity (carbon production rates as a measure of primary productivity [units of mg C/m^2/day], CbPM data for 28 months starting January 2015, http://www.science.oregonstate.edu/ocean.productivity/index.php) as well as the grid centroid's distance from the MPA boundary and the boundary of the Exclusive Economic Zone (EEZ) were added.

Random Forest Model and Partial Dependence Plots

We examined the importance of five environmental and physical (sea surface temperature, ocean productivity, depth, distance to the high seas, and distance to MPA boundary) and ten other characteristics (age, size, GDP of designating country, area/boundary length ratio, enforcement level, number of management zones, IUCN protected status category, percentage of study area located in the high seas, percentage of MPA bordering the high seas, and percent of fully or strongly protected area [as defined by Lubchenco and Grorud-Colvert (2015) (Table B.1) using a global random forest model (R package randomForest (Liaw & Wiener 2002)) following methods applied by Gill et al. 2017. Random forest models can incorporate a wide range of explanatory variables, deal with non-linear relationships between these variables and a target variable, and, as they are fully non-parametric, are considered robust regarding spatial autocorrelation of the data (Biau & Scornet 2016). The optimal random forest parameters (number of trees grown [ntree] and number of variables used to select the best split at each tree node [mtry]) were determined by considering several alternative values of these parameters and comparing them through cross-validation using the R package 'performanceEstimation' (Torgo 2014). Specifically, we compared 15 alternative parameter variants of random forests by considering values of the ntree parameter from 250 to 3,000 and values of the mtry parameter from 2 to 4. Although this is far from an exhaustive search, it is a representative sample of possible random forest variants. Trees were each fitted to a bootstrapped sample using recursive partitioning. The numbers of trees in each random forest model varied between 250 and 3,000. Random forest models were run on two levels: for fishing effort point data for each individual LSMPA using environmental and physical parameters (as described above) and for fishing effort point data for all LSMPAs combined with added socio-political parameters (see Table B.2). A separate random forest was created for five selected LSMPAs showing patterns of spillover.

Partial dependence plots were created to obtain information on the directionality of influence of each variable on fishing effort and to cross-validate random forest results using the R package 'pdp' (Greenwell 2017). These plots capture the complexity of patterns and interactions within the data and depict the influence of a variable on the response (the model prediction) after marginalizing the influence of all other variables included in the analysis (Friedman 2001). They are one of the most used tools for understanding the influence of the different variables when using models that are not directly interpretable, like it is the case of random forests.

Results

Seascapes of use around LSMPAs

Fishing pressure in a zone up to 500 km around the thirteen LSMPAs chosen as case studies was highly spatially autocorrelated (Moran's Index of 2.3 with a z-score of 490 and $p < 0.000001$) and varied by more than a factor of 10 (Fig. 4.1 B) over the three years included in this study. For example, fishing effort around Macquarie Island Marine Park (MP) in Australia was fairly low with 0.033 fishing hours/km^2, Fig. 4.2 C), while fishing intensity was high around others (e.g. Phoenix Islands Protected Area, 0.34 fishing hours/km^2 Fig. 4.2 H), compared to an overall average of 0.15 fishing hours/km^2 (median 0.08 fishing hours/km^2) as measured across all thirteen studied LSMPAs.

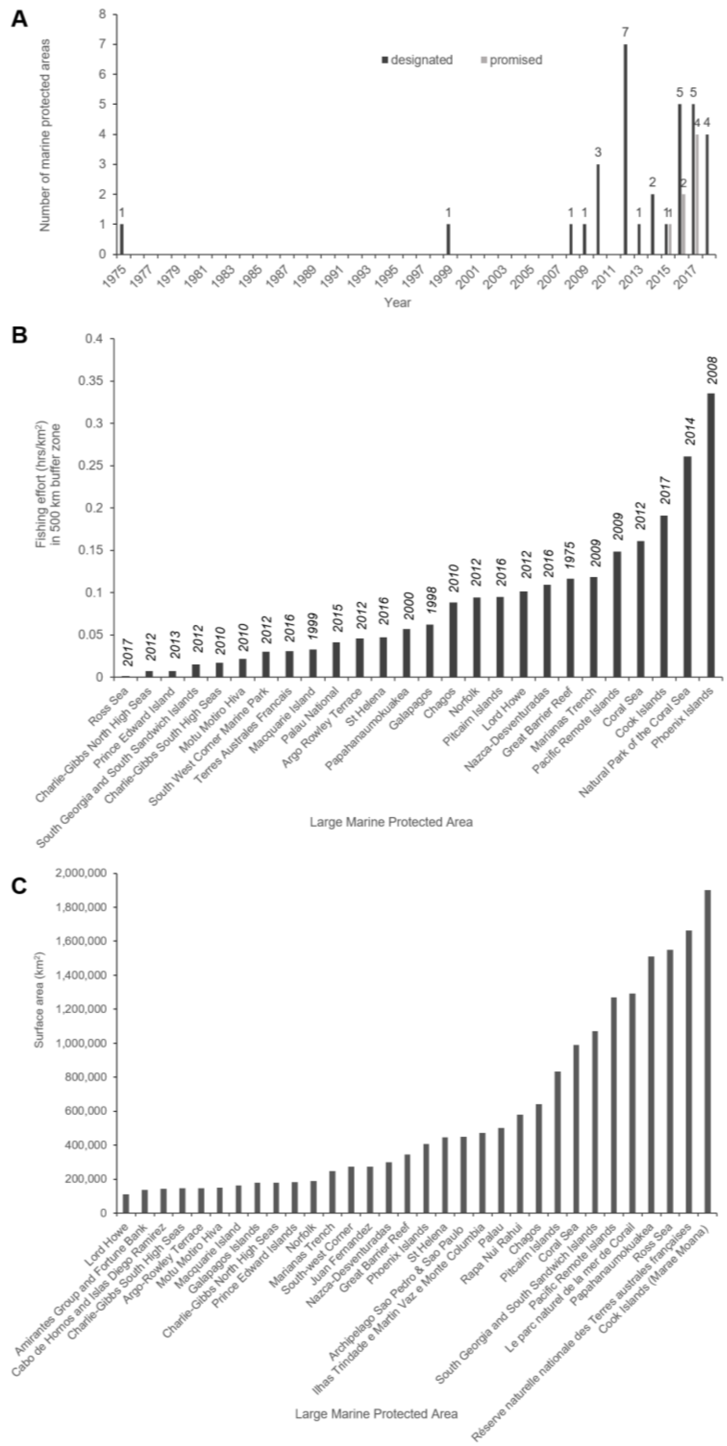
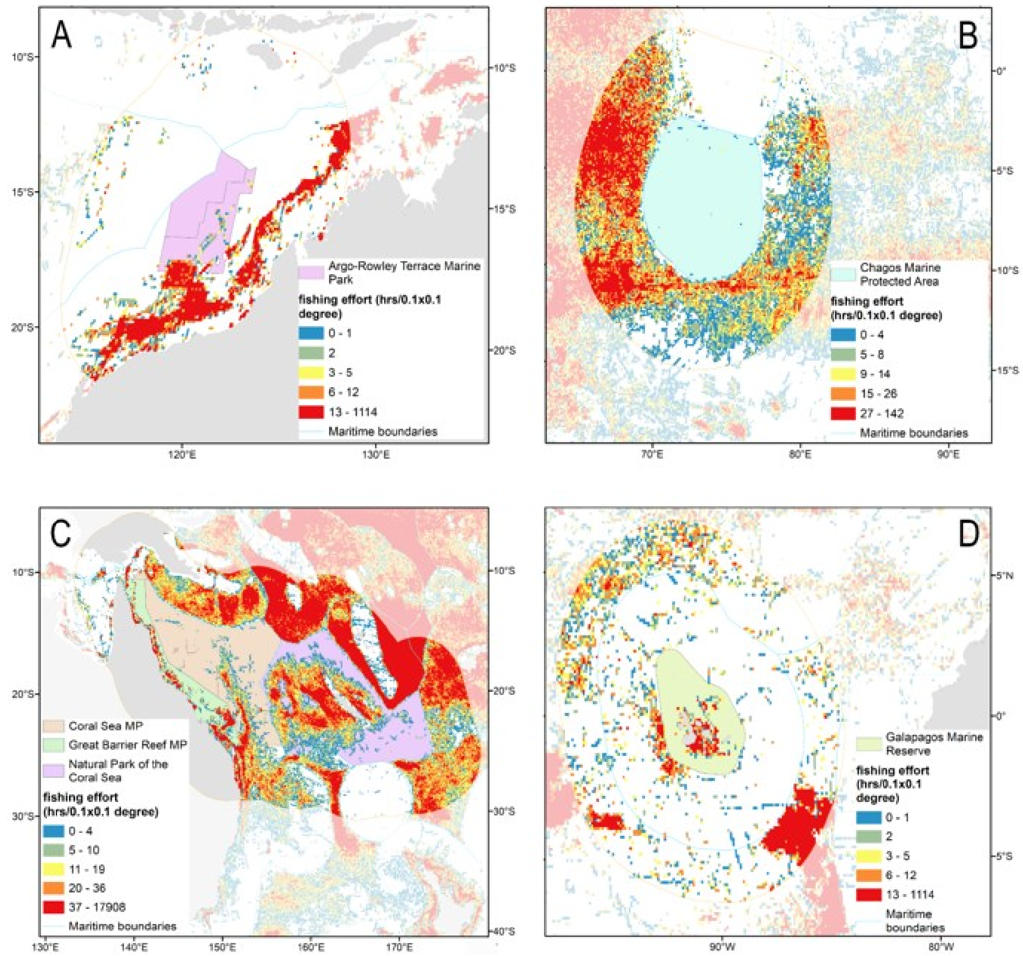


Fig. 4.1 Overview of large-scale marine protected areas (>100.000 km-2) worldwide. A: Number of LSMPAs designated (dark grey) and promised (light grey) per year since 1975. B: Fishing effort from AIS data (hours fished per km^2 in a 500 km zone around the MPAs) for established LSMPAs. C: Surface area of all LSMPAs designated by 2018.

Fishing activity within LSMPAs

Fishing activity was detected inside all LSMPAs, but overall levels summed over three years were extremely low (average of 0.06, median of 0.01 hours fished/ km^2) and only one out of the thirteen LSMPAs (Great Barrier Reef MP) had more than 0.5 fishing hours/km^2 inside the multi-zone protected area, nearly thrice the average compared to fishing effort in the 500 km zones surrounding the protected areas (0.15 hours fished/km^2).

The four LSMPAs that are fully protected (Chagos MPA, Pacific Remote Islands MNM, Papahānaumokuākea MNM and Phoenix Islands Protected Area, Table B.1)had an average of 0.007 fishing hours per km^2 inside (median 0.002 hours fished/km^2) the protected areas, more than twelve times less than the average amount of 0.089 fishing hours per km^2 (median 0.026 fishing hours per km^2) inside the remaining, partially protected nine LSMPAs.


A
Argo-Rowley Terrace Marine Park
fishing effort (hrs/0.1x0.1 degree)
0 - 1
2
3 - 5
6 - 12
13 - 1114
Maritime boundaries
B
Chagos Marine Protected Area
fishing effort (hrs/0.1x0.1 degree)
0 - 4
5 - 8
9 - 14
15 - 26
27 - 142
Maritime boundaries
C
Coral Sea MP
Great Barrier Reef MP
Natural Park of the Coral Sea
fishing effort (hrs/0.1x0.1 degree)
0 - 4
5 - 10
11 - 19
20 - 36
37 - 17908
Maritime boundaries
D
Galapagos Marine Reserve
fishing effort (hrs/0.1x0.1 degree)
0 - 1
2
3 - 5
6 - 12
13 - 1114
Maritime boundaries

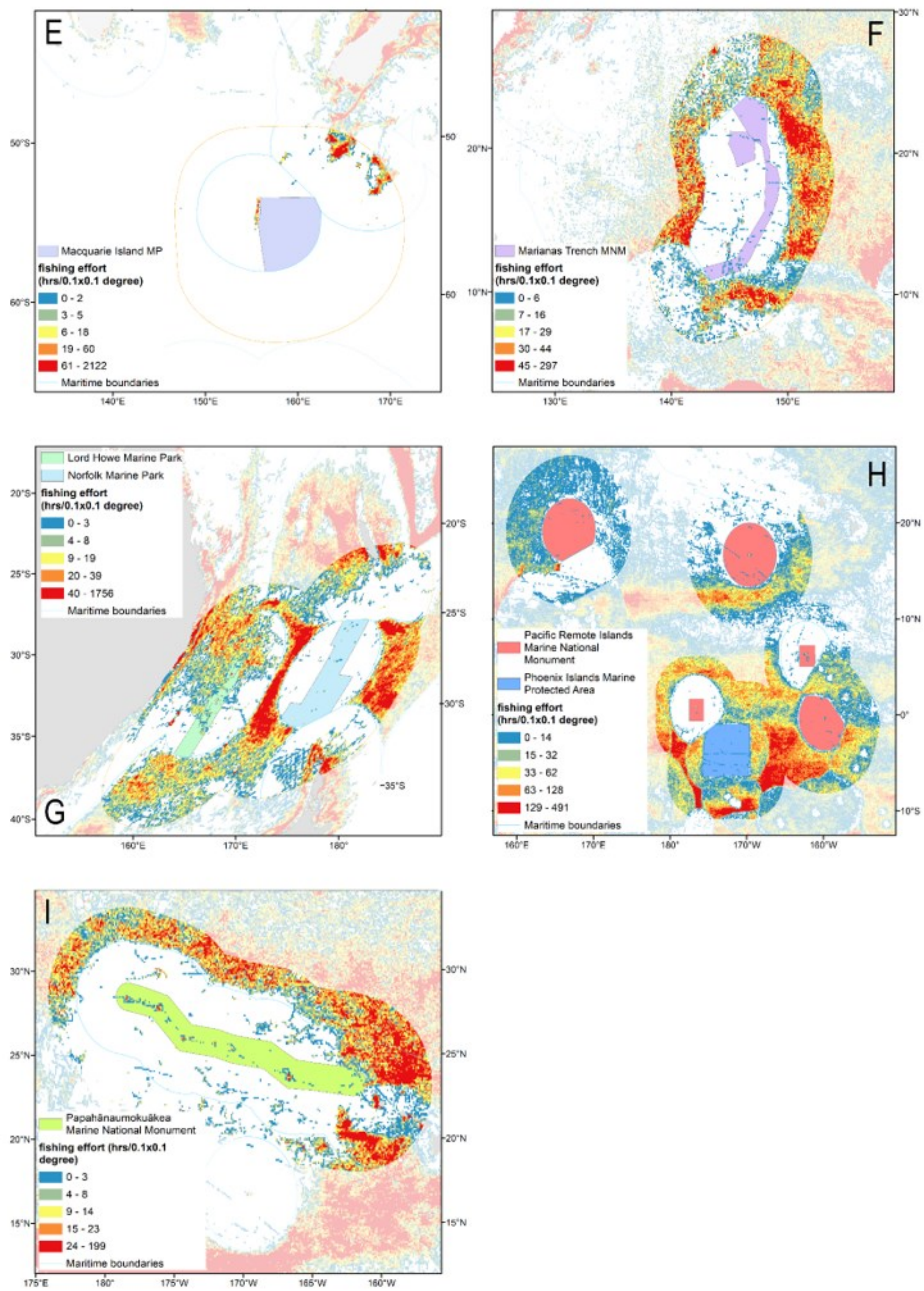


Fig. 4.2 Spatial distribution of fishing effort around thirteen selected large-scale marine protected areas (LSMPAs). Fishing effort summed for years 2015 to 2017 and broken into quantiles. Transparent areas denote the fishing activity study areas, extending 500 km from LSMPA boundaries. Land is shown in grey, white denotes ocean area with missing data. Only LSMPAs established before 2015 and with fishing effort exceeding 0.031 fishing hours/km^2 were chosen. Note that maximum values are given for each individual LSMPA or highest value where buffer zones around LSMPAs overlap. MP denotes Marine Park, MR Marine Reserve, and MNM Marine National Monument. Map data from Natural Earth.

Effects of distance from MPA boundary

Patterns of fishing intensity with distance from the MPA boundary varied greatly for each of the thirteen case studies (Fig. 4.3). On average, fishing intensity increased gradually with increasing distance from the MPA. This pattern, however, was heavily influenced by the increasing proportion of high seas area (i.e. area with more open fisheries access) covered when moving further away from MPA boundaries. Within EEZs, average patterns of fishing effort as a function of distance to MPA boundaries were less pronounced. Areas further away from MPA boundaries still featured the highest average fishing effort as fishing effort located in other, neighboring EEZs was included (Fig. B.1). Contrary to this general trend, fishing effort appeared to aggregate close to the boundaries of five MPAs: Argo-Rowley Terrace MP, Coral Sea MP, Galápagos Marine Reserve (MR), Great Barrier Reef MP and Macquarie Island MP. It is interesting to note that Galápagos MR, Great Barrier Reef MP and Macquarie Island MP are also the three oldest LSMPAs included in this study (Table 4.1), and the only ones designated before 2000. This pattern might be indicative of built-up fish biomass spilling over from the protected area, and adjacent fisheries capitalizing on this, as documented previously for the Galápagos MR (Boerder et al. 2017).

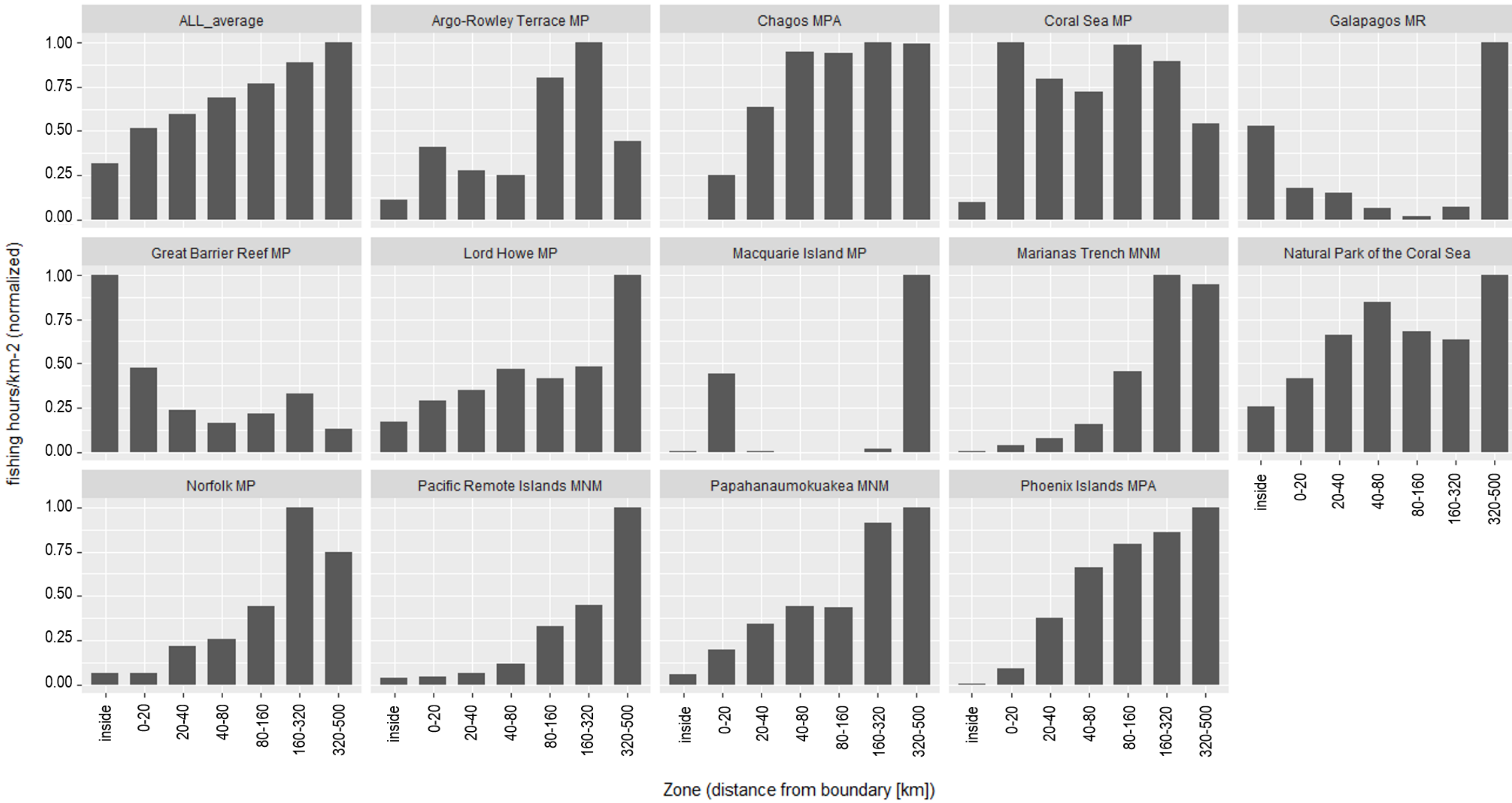


Fig. 4.3 Effects of distance from protected area boundary on fishing effort. Fishing effort is expressed as the number of hours fished per km^2 within a 500 km radius from MPA boundary and normalized to 1. Note that the study areas outside of the LSMPAs encompass areas in Exclusive Economic Zones as well as in the high seas. MP denotes Marine Park, MR Marine Reserve, and MNM Marine National Monument.

Five LSMPAs with potential spillover effects were investigated in more detail using fine-scale (0.01 by 0.01 degree grid) fishing effort data in the immediate vicinity of the MPA boundary. For these five LSMPAs, more fishing effort was indeed found to concentrate within the first approx. 10 km from the boundary (Fig. B.2).

Factors predicting fishing intensity around LSMPAs

When exploring the relative importance of five physical predictor variables around thirteen individual LSMPAs, no single predictor variable explained the distribution of fishing intensity around all individual LSMPAs (Table 4.2). Distance to the MPA boundary was the most important factor for seven out of the thirteen LSMPAs, followed by distance to the high seas (Table 4.2). Fishing effort was seen to increase further away from MPA boundaries and was higher closer to, as well as in the high seas. Sea surface temperature, ocean productivity (carbon production rate) and depth had less influence.

Table 4.2 Relative importance of six variables in explaining the distribution of fishing effort around thirteen large-scale marine protected areas (LSMPAs). Mean decrease accuracy (percent increase of the mean squared error, %IncMSE) of selected predictor variables and percentage of variance explained as predicted by random forest models for each individual LSMPA: overall percentage explained by model (%VAR_EXPL) as well as the percentage of the mean squared error (%IncMSE) of: Sea surface temperature (SST), ocean productivity (PRODUCTIVITY, carbon production rates), depth (DEPTH), the distance to the high seas (DISTANCE HS) and to the MPA boundary (DISTANCE MPA). The single most important predictor variable in each case is highlighted in bold. Note that the magnitude of %IncMSE values depend on the individual model and cannot be compared between models. MP denotes Marine Park, MR Marine Reserve, and MNM Marine National Monument.

MPA	*VARIANCE*	SST	PRODUCTIVITY	DEPTH	DISTANCE HS	DISTANCE MPA
Argo-Rowley Terrace MP	*30.7*	33.5	36.1	19.7	**38.3**	32.4
Chagos MPA	*55.6*	128.9	85.4	84.8	27.1	**214.6**
Coral Sea MP	*34.3*	21.3	11.5	15.2	17.9	**28.9**
Galápagos MR	*54.4*	**53.4**	37.1	14.9	6.1	36.9
Great Barrier Reef MP	*20.3*	17.4	20.5	14.8	**22.1**	11.7
Lord Howe MP	*63*	**69.8**	36.9	30.3	41.5	58.8
Macquarie Island MP	*42.8*	28.5	26.4	10.2	19.7	**31.98**
Marianas Trench MNM	*49.1*	106.4	78.5	41.2	106.3	**163.2**
Natural Park of the Coral Sea	*64.6*	10	20.9	26.6	9.9	**27.3**
Norfolk MP	*38.7*	12.3	16.3	22.4	13.5	**35.1**
Pacific Remote Islands MNM	*81.3*	152	253.8	82.3	**295.4**	209.2
Papahānaumokuākea MNM	*50.4*	74.8	**114.3**	47.7	59.1	107.1
Phoenix Islands PA	*83.8*	134.2	98.4	63.8	179.2	**181.6**

A closer examination of fishing effort around the immediate boundaries of the five LSMPAs exhibiting spillover pattern using a random forest model revealed that overall patterns for these five LSMPAs are mostly correlated to maritime zoning, sea surface temperature, as well as ocean productivity (Fig. B.3).

When testing all 15 environmental, physical, and additional characteristics across all LSMPAs combined (38.23 % variance explained, 1,000 trees grown), maritime zoning had the strongest influence on the distribution of fishing effort with the distance to the high seas (18.6% mean decrease accuracy) and to the MPA boundary (11.1% mean decrease accuracy) as well as MPA size (10.3% mean decrease accuracy) and area of the buffer zone located in the high seas (7.8% mean decrease accuracy) being the top major predictor variables (Fig. 4.4).

As evident from the maps of fishing intensity (Fig. 4.2) as well as the analysis of fishing effort as a function of distance to the MPA boundary (Fig. 4.3), fishing effort is highest closer to and in the high seas. Therefore, distance to the high seas and the distance to the MPA boundary can be considered as correlated. However, when changing the parameter 'distance to the high seas' from a continuous variable to a discrete variable (fishing located in the high seas yes/no) the importance of this predictor variable increased further (27.7% mean decrease accuracy, 30.48% variance explained by random forest model).

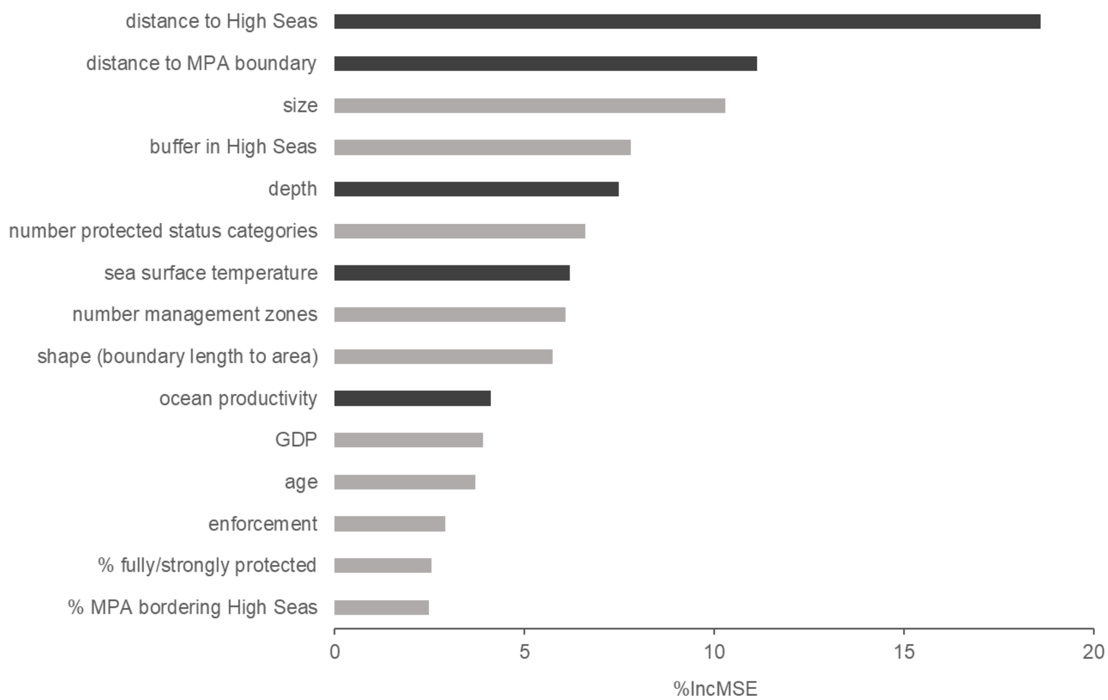


Fig. 4.4 Variable importance in predicting patterns of fishing effort around thirteen large-scale marine protected areas (LSMPAs) from a random forest model. Environmental and physical parameters are dark grey, additional parameters are light grey. %IncMSE (percent increase of mean squared error) denotes the mean decease in accuracy of the model if the respective parameter is dropped from the model. Higher values indicate greater predictive capacity of variables.

The majority of variables showed a clear directionality and pattern of their influence on fishing effort (Fig. 4.5). Most fishing effort around LSMPAs took place in waters either below 16 °C or above 21 °C, in shallower, more productive waters (as measured by carbon production rates), coastal areas (furthest away from high seas) and outside of EEZs in the high seas. Regarding relations to the LSMPAs, more fishing effort overall was apparent around younger MPAs and bigger MPAs, as well as around LSMPAs designated by countries with lower GDP, and with low enforcement. Fishing effort seemed to be higher around LSMPAs with more fully or strongly protected area but fewer overall individual zones.

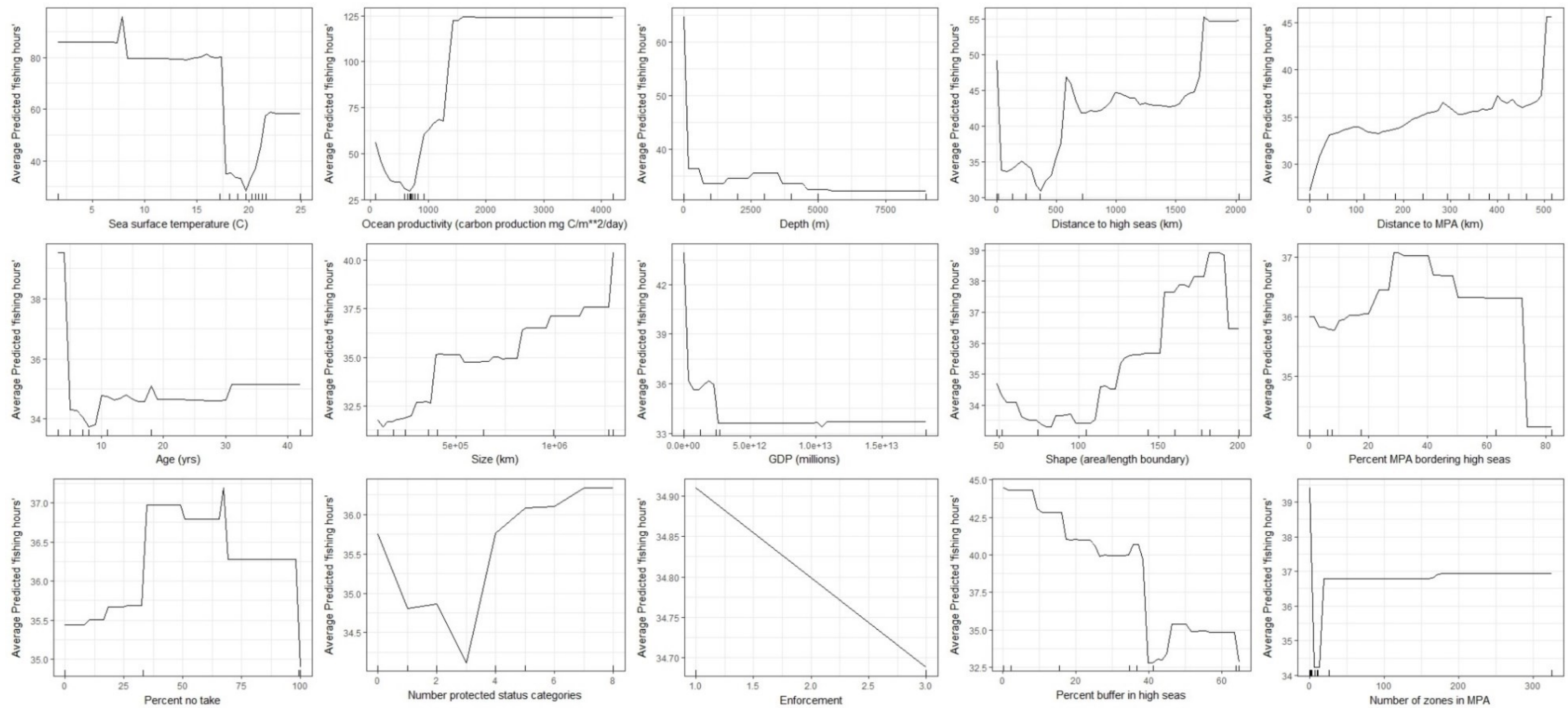


Fig. 4.5 Partial dependence of fishing effort (fishing hours) on each variable as predicted by a random forest model. Depicted is the influence of one predictor variable onto the target variable (model prediction, here: fishing effort) after controlling for the influence of all other variables included.

Discussion

Designated and promised LSMPAs now account for about 7% of global ocean area, with a combined surface area nearly 20 times larger than the rest of all MPAs combined. Despite this spatial dominance, due to the recent acceleration in the establishment of LSMPAs the majority of the research of interactions between fisheries and protected areas have focussed on smaller, coastal MPAs. Such studies found that large size and age, no-take regulations, remote location, as well as proper enforcement, funding and staffing influence the effects of MPAs on biodiversity and fisheries (Claudet et al. 2008, Edgar et al. 2014, Friedlander et al. 2017, Gill et al. 2017). Here we find that fishing activity around the worlds' largest MPAs was mainly influenced by maritime zoning regulations with distance to MPA boundaries and to the high seas of primary importance in shaping fishing effort around individual LSMPAs. While fishing effort tended to peak in high seas areas away from individual MPAs, older MPAs also showed some evidence of close aggregation of fishing effort with the immediate MPA boundaries, indicative of local spillover effects. Overall fishing intensity also tended to be higher around younger, larger LSMPAs in temperate, shallower, and more productive waters, designated by countries with lower GDP and potentially less enforcement capabilities (Fig. 4.5). Fully protected LSMPAs had considerably less observed fishing effort inside the protected areas than partially protected ones (twelve times less). Considering of these variables could guide decisions of siting as well as management of existing and future LSMPAs.

The overriding effect of MPA placement with respect to Exclusive Economic Zones (EEZs) and the high seas, respectively, represents their long-standing establishment and importance: EEZs came formally into force under the United Nations Convention on the Law of the Sea (UNCLOS) in 1982. The effectiveness of many EEZ boundaries in regulating fisheries is obvious when examining patterns of fishing effort, for example, around the Pacific islands and US overseas territories surrounded by high intensity of fishing effort in the high seas but virtually devoid of (visible) fishing effort (Fig. 4.2) (see also global pattern documented by Kroodsma et al. 2018). This observation reinforces suggestions that fisheries management within these EEZs may provide a valuable and desirable compliment to reinforcing the same outputs that are sought from LSMPAs. Such well-enforced EEZs may present opportunities for future conservation efforts, as they would not displace much fishing effort and might help to lock a low footprint relative to the intense use in surrounding high seas. An example is the Pacific Remote Islands Marine National Monument (MNM), which encompasses EEZs of several US overseas territories. This also stresses the value of calls for improved management and environmental protection in existing EEZs as a necessary complement to the designation of LSMPAs (De Santo 2013).

Geographic placement of LSMPAs has been the subject of valuable debate amongst marine conservation strategists. While some argue for meaningful protection of near-pristine remote places to safeguard against future risks and unsustainable exploitation (McCauley et al. 2013), others have reasoned that the establishment of LSMPAs in remote areas with low fishing pressure (e.g. areas dubbed 'residual reserves') does a disservice to marine conservation for these regions already enjoy de-facto protection (Craigie et al. 2014, Devillers et al. 2014). The argument being made is that the limited resources available to spend on conservation measures are better placed in areas experiencing heavy use and degradation (Bottrill et al. 2008). Furthermore, the preferential designation of overseas territories as MPAs by nations largely located in the Global North has drawn critique as 'outsourcing' of environmental protection, sometimes at the costs of the local population such as in the case of the British Indian Ocean Territory/Chagos MPA (De Santo et al. 2011, O'Leary et al. 2018).

This analysis shows that this is not at least the case for all LSMPAs. Our analysis revealed high levels of fishing effort around (and sometime within, as below) some LSMPAs included in the sub-set examined in this study. Fishing effort, for example, around the Phoenix Islands Protected Area in Kiribati in the Central Pacific was measured at 0.34 fishing hours/km^2, which is ten times higher than in areas experiencing low fishing effort such as around Macquarie Island MP, an uninhabited island south of Australia. While there remains a diversity of opinions regarding the advantages and disadvantages of situating LSMPAs in regions of intense fishing effort, we hope these analyses demonstrate how AIS data can constructively be employed to quantitatively measure fishing effort in contexts of contemporary and future LSMPAs. This insight may be especially useful in more systematic conservation planning especially for large-scale protection in areas beyond national jurisdiction (Agardy et al. 2011, Ban et al. 2014, Davies et al. 2017).

While a 'fishing the line' behavior, a potential indicator of spillover of fish from the protected area, has been observed for smaller reserves and closures (e.g. Murawski et al. 2000, Follesa et al. 2011, Da Silva et al. 2015, Tewfik et al. 2017), only a few studies have examined this process for LSMPAs, concentrating on selected fisheries (Boerder et al. 2017, Bucaram et al. 2018). While the analysis across all thirteen LSMPAs presented here did not indicate general fishing the line behavior, fishing effort seemed to accumulate along the boundaries of five LSMPAs, including the three oldest ones (Galápagos MR, Great Barrier Reef MP, Macquarie Island MP). Age has been found to be important in MPA effectiveness before (Edgar et al. 2014) and our observations suggest that maturity of protected areas might affect surrounding fisheries at least in some cases.

However, the variety of patterns observed in this study highlights the diversity of factors influencing fisheries around LSMPAs and the importance of their unique settings. For example, the Great Barrier Reef MP is heavily zoned with a number of multiple-use zones allowing for certain types of fishing inside the marine park (Day 2002) and explaining the higher than average fishing effort in protected vs unprotected areas. It also has a history of poaching mainly by recreational fishermen (Bergseth et al. 2017). Both factors influence patterns of fishing effort within and outside the protected area (Fig. 4.2). Additionally, the Great Barrier Reef MP is directly adjoined by the Coral Sea MP, further influencing fishing patterns along its boundaries (Fig. 4.2). Data for the Galápagos MR were influenced by a strong El Niño event in 2015/2016 which is known to negatively affect fisheries in the Eastern Tropical Pacific (Worm et al. 2005). This might explain the lower magnitude of fishing effort aggregation and difference of patterns as observed for the marine reserve previously (Boerder et al. 2017). Macquarie Island MP shows a weak spillover effect, however, observed fishing effort around the marine park is very sparse and driven by about 63 vessels over three years (Fig. 4.2). In this context it would be of interest to repeat this analysis for these and other LSMPAs on a decadal scale in the future as well as improve vessel monitoring to capture fishing effort by vessels not equipped with AIS (see discussion below).

Additionally, characteristics of location and management of individual LSMPAs are likely to influence the pattern of fishing effort around their boundaries. Next to factors mentioned such as multi-use zoning allowing fishing inside some areas of the Great Barrier Reef, local characteristics such as currents and proximity to areas of high fishing intensity such the main Hawaiian Islands in relation to the Papahānaumokuākea MNM are likely affecting the analysis of fishing effort intensity in relation to MPA boundaries (Fig. 4.3).

A variety of additional reasons could explain the lack of a general pattern for the other LSMPAs: spillover might be 1) masked by other processes, 2) not yet visible due to young age of most LSMPAs, 3) overridden by insufficient enforcement, or 4) hampered by management capacity shortfalls. Here we examine these hypotheses in more detail.

On a local scale, a variety of environmental and oceanographic factors might influence fishing patterns and potentially mask any MPA effects, such as benthic and pelagic features (depth, thermal fronts, currents), weather, location (distance to land or port) and others (see below). Additionally, most LSMPAs are located in remote, open ocean areas mostly utilized by highly migratory pelagic target species such as tuna, billfishes, and sharks. While some of these species follow specific migration routes (i.e. some sharks and bluefin tunas), others are opportunistic and migrate following preferred environmental features such as thermal fronts. These species therefore might or might not be present within or around a particular MPA in any given year, resulting in highly variable patterns of fisheries targeting these species. To better resolve potential spillover effects for these species (Swain & Wade 2003) more localised studies focussed on particular target species, their biology, and the associated fishery may be necessary (Sibert et al. 2012).

Spillover patterns might also be affected by the age of the MPAs – spillover effects are generally thought to develop after enforcement has actually altered fishing patterns for a number of years and the target species had enough time to respond, given its life history (Barrett et al. 2007, Babcock et al. 2010). For coastal MPAs and shallow water species MPA effects are thought to develop after about 5 years (Babcock et al. 2010) whereas for pelagic migratory species these might develop over one to two generations of the target species, which may be 15 years and more e.g. for bigeye tuna (*Thunnus obesus*) (Sibert et al. 2012, Moffitt et al. 2013). Ten out of the thirteen case studies included in this analysis are younger than that (before 2015, start of AIS data included in this analysis), only the Great Barrier Reef MP (1975), the Galápagos MR (1998) and the Macquarie Island MP (1999) are older. Interestingly, all three appear to attract fishing effort to their boundaries.

A scarcity of apparent spillover patterns might also be caused by insufficient enforcement of the MPA (Cressey 2011, Dulvy 2013). This, however, does not seem to be a major factor as most LSMPAs observed in this study had little detectable fishing effort inside (Fig. 4.2). Remarkably, fully protected LSMPAs had about twelve times less fishing effort inside the protected area than partially protected LSMPAs, such as the heavily zoned Great Barrier Reef MP, which displayed nearly thrice as much fishing effort inside the protected area than outside. While most of this fishing effort was probably within multi-use zones permitting fishing, enforcement within complex zoning plans can become a challenge. Problems with integrated zoning as well as enforcement have been highlighted as key issues for the marine park before (Jones et al. 2011). The fact that fishing intensity appeared to be higher around LSMPAs with weaker enforcement potentially further influences MPA effects and emphasizes the importance of management. However, as not all fishing vessels are equipped with AIS it is likely that especially illegal fishing effort inside the LSMPAs was not detected (Kroodsma et al. 2018).

Our findings differ from previous analyses evaluating factors of MPA effectiveness: three studies found age, size and no-take area to be important factors influencing efficiency of MPAs (Claudet et al. 2008, Edgar et al. 2014, Friedlander et al. 2017), whereas management capacity (number of staff, budget) was most influential in another (Gill et al. 2017). However, these studies examined a different response variable, looking for MPA effects on biomass inside and outside the protected areas, and largely only included small, coastal MPAs. As coastal MPAs experience quite a different set of influences compared to MPAs located further offshore, parameters examined in these studies differ, i.e. do not include maritime zoning as this is unlikely to affect an MPA located far away from EEZ borders. Thus, it is not surprising that we find maritime zoning to play an important role for LSMPAs, indicating that their unique setting needs to be taken into account for adequate design and management.

Given the importance of maritime zoning and fishing effort in the high seas, we suggest that special consideration should be given to areas where the MPA borders the high seas when designing and enforcing LSMPAs, as these boundaries appear to be the most vulnerable regarding attraction of effort and potentially also infringements. A possible solution would be to draw MPA boundaries a short distance before the EEZ border so the EEZ serves as a buffer zone for the MPA while providing national fisheries with exclusive access to potential spillover benefits. Furthermore, LSMPAs could be created in the high seas to enhance spatial protection within EEZs by shielding EEZ boundaries from high fishing intensity. The influence of the border to the high seas is also important to consider when designating different zones, i.e. no-take zones, as well as for future planned MPAs in the high seas. For these cases, a buffer zone around the MPA or its' no-take zones seems a solution to counteract boundary infringements as well as to buffer high fishing intensity along the boundaries. Furthermore, as we found that the age of LSMPAs appears to influence the attraction of fishing effort to protected area boundaries, management plans should consider dealing with increasing future fishing effort outside MPA boundaries as it has the potential to negate any MPA benefits regarding enhanced biomass export (Walters 2000, Goñi et al. 2008).

There are several limitations of this study: first and foremost, AIS does not capture all fishing effort as vessels that do not carry an AIS transponder, switch it off, or tamper with it otherwise, will not be picked up or their signal might be wrongly interpreted (Kroodsma et al. 2018). Furthermore, vessel type detection, based on movement patterns, might misidentify vessel types, i.e. identify a cruise ship as a fishing vessel. The extent to which misclassification of vessel types is happening is impossible to estimate due to a lack of comprehensive fishing vessel databases for comparison. Lastly, AIS coverage varies greatly due to differing national regulations, and is generally better for larger vessels (Kroodsma et al. 2018). Thus, a lot of fishing activity might go undetected especially in areas where smaller vessels are active, such as the Great Barrier Reef MP.

We did not consider a range of potential factors influencing fishing effort, mostly due to a lack of global data in sufficient fine scale. Further possible predictors of fishing effort include (but are not limited to): weather, currents, fuel price, distance to port, distribution of biodiversity, oceanographic features such as fronts, types of fishing (i.e. use of fish aggregation devices), as well as other factors such as management capacity (number of staff, budget), management by Regional Fisheries Management Organizations, and fisheries agreements.

We anticipate that this work is just the foundation of extensive future research specifically dedicated to exploring characteristics and effects of large MPAs, given their spatial dominance in marine conservation coverage globally. Next steps along this road potentially include the analyses of additional factors as mentioned above, as well as repetitions of this study at longer time intervals such as decadal scales to monitor changes associated with maturing MPAs. Additionally, with improving vessel monitoring data, smaller fleets should be included especially in the monitoring of coastal LSMPAs such as the Great Barrier Reef MP or those surrounding populated islands such as the Galápagos MR.

Conclusion

Understanding the effects of LSMPAs on surrounding fisheries, and vice versa, is important to properly assess the effectiveness and management needs of these relatively novel form of MPAs. Modern monitoring and surveillance tools such as AIS can help observe fisheries even in remote waters and over large areas, in near-real-time, and as such offer the only realistic solution for LSMPA assessment and monitoring, although enforcement still remains a challenge. While we are just starting to study and understand the potential of large-scale marine conservation, and the majority of global LSMPAs are still fairly young, evidence of effects of LSMPAs are already emerging. The attraction of fishing effort to the oldest LSMPAs found in this study showed the potential importance of maturation of MPAs. However, the fact that EEZs have an even stronger effect on fishing effort patterns than MPAs indicates the significance of global maritime zoning, as well as the length of time this zoning has been established. It also highlights the importance of pairing MPA establishment with strong fisheries management practices within these regulated EEZ waters. Given these patterns and the observed low levels of fishing effort inside the LSMPAs, the success of EEZs and their effect on the distribution of fishing effort hold promise that marine spatial protection can be successfully implemented even at large scales. While conservation potential of EEZs could be improved through more focus on conservation issues in fisheries management within national waters, the lessons learnt from EEZ management and enforcement might serve to guide establishment of LSMPAs even in areas beyond national jurisdiction.

Appendix B

Table B.1 Marine protected area (MPA) characteristics selected for the study.

MPA Characteristic	Units	Source
Age	Age of LSMPA before 2017 (years)	Calculated: (2017 - 'Year designated')
Size	Area of LSMPA (km^2)	Updated for each LSMPA from O'Leary et al. 2018
Gross Domestic Product (GDP)	Average GDP (2015-2016) of designating country (US$)	https://data.worldbank.org/indicator/NY.GDP.MKTP.CD (accessed 25/06/2018)
Shape	Ratio between LSMPA area and boundary length (dimensionless)	Calculated in ArcGIS using the Winkel Tripel projection
Percent of LSMPA bordering the high seas	Percentage of LSMPA boundary that borders the high seas (percent)	$\left(\frac{\text{Length of MPA boundary bordering the High Sea}}{\text{Length of MPA boundary}}\right) * 100$ MPA shapefiles obtained from the World Database on Protected Areas
Percent no take	Proportion within the protected area where no fishing is allowed (percent)	Updated for each LSMPA from O'Leary et al. 2018
Number of protection categories	Number of different protection classifications used within LSMPA zoning (e.g. IUCN/regional categories) (number)	Based on available management plans and supplementary online information for each LSMPA.
Enforcement	Scaled 1-3: 1 weak, 2 intermediate, 3 strong	Estimated from authors knowledge

MPA Characteristic	Units	Source
Percent of buffer zone in the high seas	Proportion of study area (500 km buffer from MPA boundary) located in the high seas (percent)	Calculated in ArcGIS using the Winkel Tripel projection MPA shapefiles obtained from the World Database on Protected Areas
Number of management zones	Number of distinct management zones within the LSMPA (number)	Based on available management plans and supplementary online information for each LSMPA

Table B.2 Characteristics of marine protected areas (MPAs) included in this study. Status as of 2015.

MPA	age	size (km-2)	GDP (million)	shape (area/ boundary)	% MPA bordering high seas	% fully/strongly protected	number protected status categories	enforcement category	% buffer zone in high seas	number zones in MPA
Argo-Rowley Terrace MP	5	146,003	1,275,000	74.9	19	0	3	1	18.0	4
Chagos MPA	7	639,661	2,739,988	200.4	81.9	100	1	1	64.7	1
Coral Sea MP	5	989,924	1,275,000	177.4	0	0	4	1	2.2	26
Galapagos MR	19	138,000	98,990	79.3	0	0	0	2	41.0	1
Great Barrier Reef MP	42	348,700	1,275,000	48.7	0	33	8	3	0.3	325
Lord Howe MP	5	110,139	1,275,000	54	6.2	0	5	1	41.3	7
Macquarie Island MP	18	162,000	1,275,000	90.7	36.7	36	2	2	48.2	3
Marianas Trench MNM	8	246,608	18,302,874	52.2	4	17	2	1	34.6	3
Natural Park of the Coral Sea	3	1,291,000	2,449,500	182.3	17.5	0	0	1	15.6	0
Norfolk MP	5	188,444	1,275,000	74.3	7.5	0	2	1	33.7	2
Pacific Remote Islands MNM	18	1,269,094	18,302,874	143.8	63.2	100	2	1	63.8	10
Papahānaumokuākea MNM	17	362,061	18,302,874	104.8	0	100	3	3	43.9	11
Phoenix Islands PA	11	408,250	163	160.3	0	99.4	2	1	36.8	2

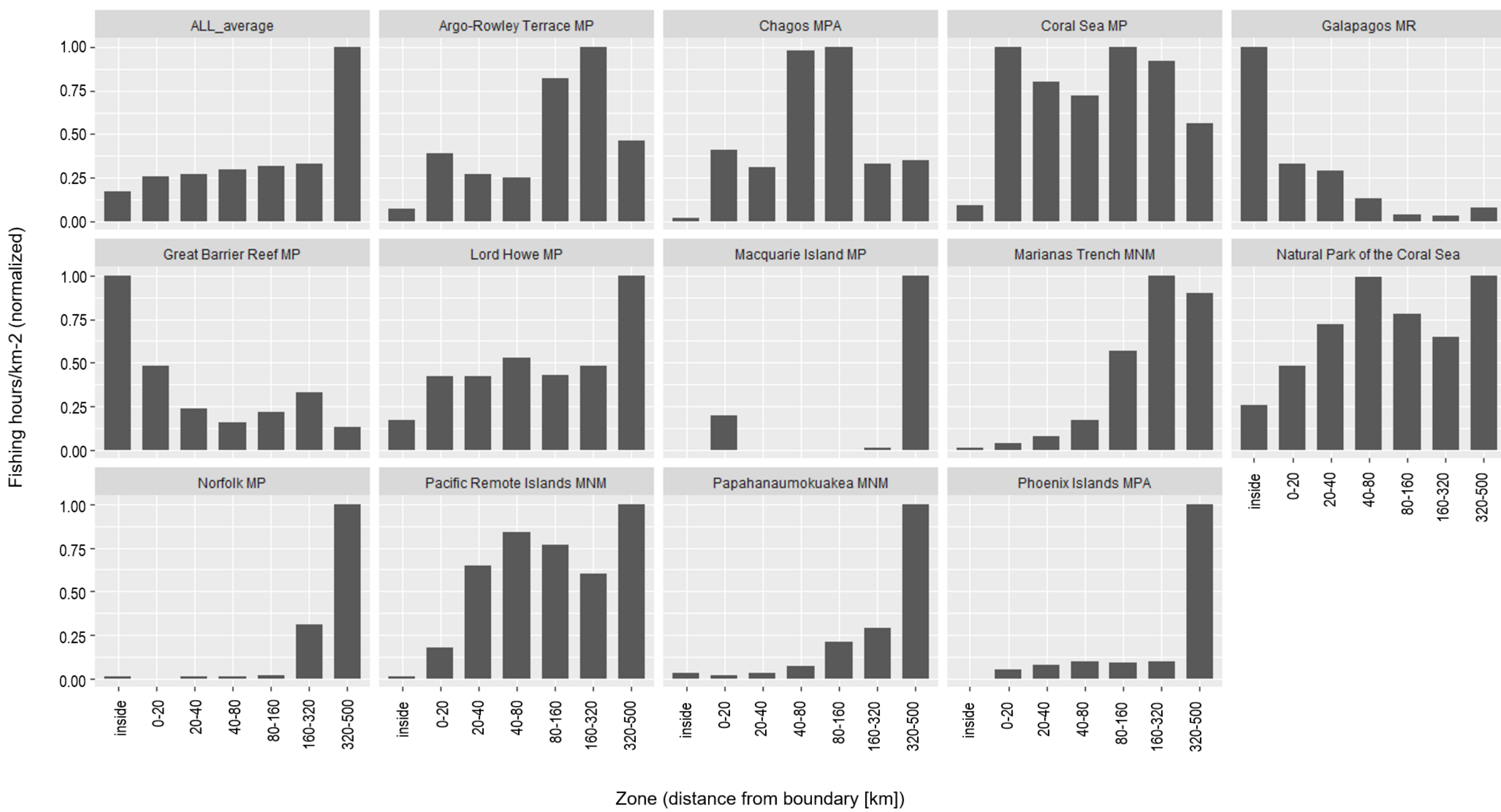


Fig. B.1 Fishing effort within Exclusive Economic Zones surrounding large-scale marine protected areas. All high seas areas were excluded from analysis.

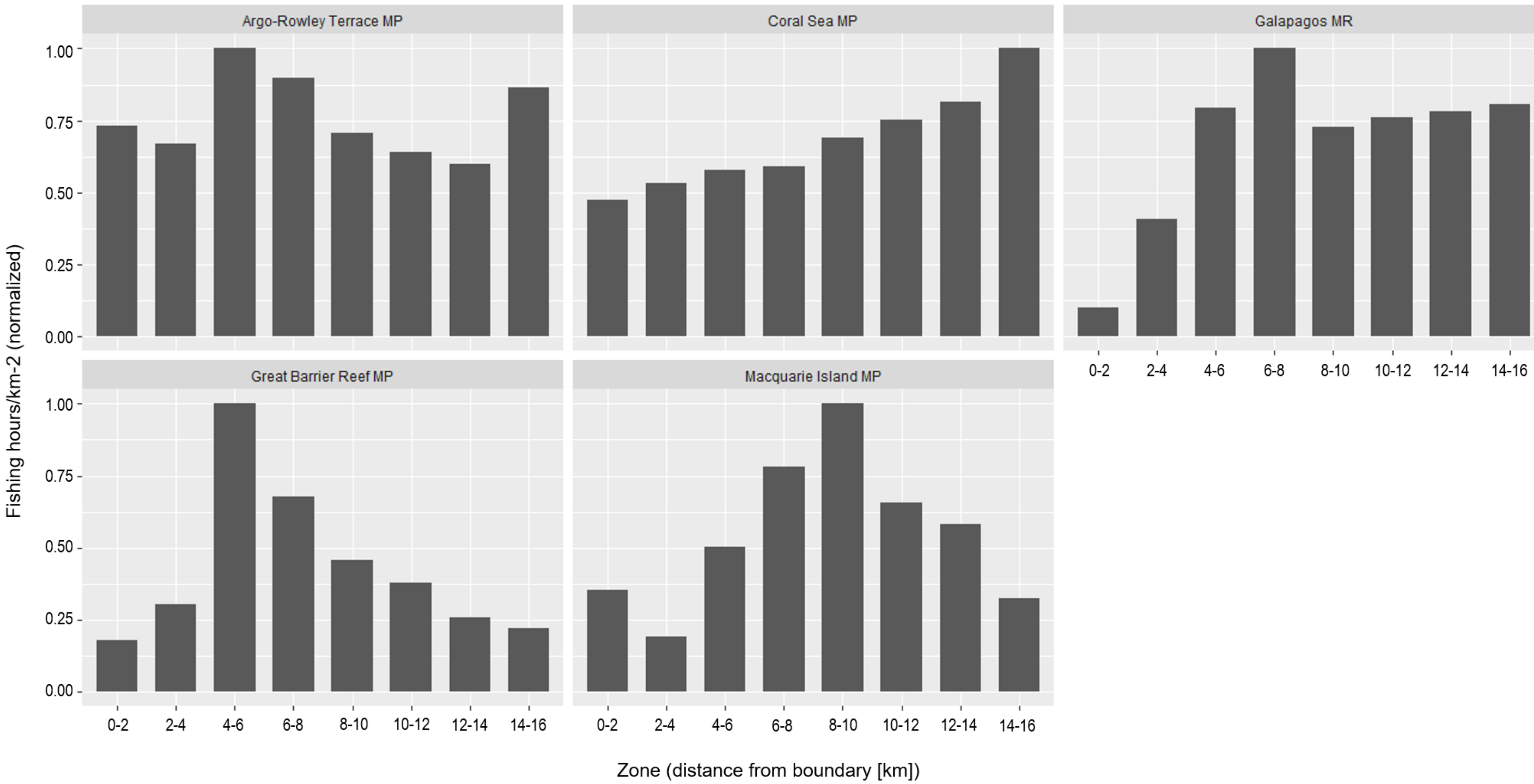


Fig. B.2 Fishing effort (hrs/km^2) with distance from marine protected area boundary for five selected case studies showing spillover patterns. Data normalized to 1.

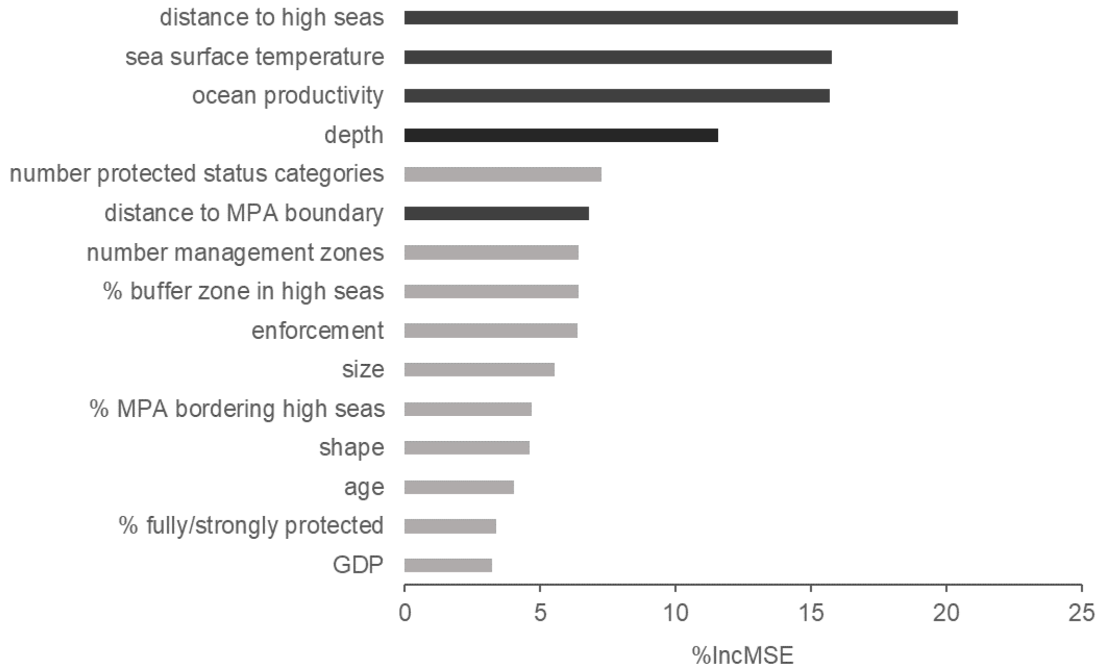


Fig. B.3 Random forest variable importance in predicting patterns of fishing effort around the five large-scale marine protected areas (LSMPAs) showing patterns coincident with spillover. Environmental and physical parameters varying for each datapoint given in dark grey, additional parameters varying for each MPA in light grey. %IncMSE (percent increase of mean squared error) denotes the mean decrease of accuracy of the model if the respective parameter is dropped from the model.

CHAPTER 5
GLOBAL HOT SPOTS OF TRANSSHIPMENT OF FISH CATCH AT SEA

Abstract

A major challenge in global fisheries is posed by transshipment of catch at sea from fishing vessels to refrigerated cargo vessels, which can obscure the origin of the catch and mask illicit practices. Transshipment remains poorly quantified at a global scale, as much of it is thought to occur outside of national waters. We employed Automatic Identification System (AIS) vessel tracking data to quantify spatial patterns of transshipment for major fisheries and gear types. From 2012-2017 we observed 10,510 likely transshipment events, with trawlers (53%) and longliners (21%) involved in a majority of cases. Trawlers tended to transship in national waters whereas longliners did so predominantly on the high seas. Spatial hot spots were seen off Russia, West Africa and in the South Indian and equatorial Pacific Ocean. Our study highlights novel ways to trace seafood supply chains and identifies priority areas for improved trade regulation and fisheries management at the global scale.

Introduction

Seafood is the world's most traded food commodity with global exports worth more than US$148 billion in 2014 (FAO 2016). The vast majority of fish and shellfish (78%) is processed and traded internationally through complex supply chains that connect fishing vessels with individual consumers (FAO 2016). Most of the global catch estimated at 100 million metric tons year^{-1} (Pauly & Zeller 2016) is landed directly by fishing vessels in port, particularly from vessels that operate closer to the coast and in national waters. Larger fishing vessels

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Boerder, K., Miller, N.A., Worm, B., 2018. Global hot spots of transshipment of fish catch at sea. Science Advances 4, eaat7159

and those fishing further offshore and on the high seas, however, often offload catch to refrigerated cargo vessels (‘reefers’) instead, while often also being resupplied with food, water, bait, crew and fuel; this common practice is known as transshipment of catch at sea (hereafter referred to as ‘transshipment’).

It has been previously reported that the majority of species subject to transshipment are high-seas related species such as tuna, sharks and billfishes (Gianni & Simpson 2006), but other species including groundfish, salmon, and crustaceans also get transshipped in both national and international waters (Ewell et al. 2017). Transshipment increases the efficiency of fishing by eliminating trips back to port for fishing vessels while maintaining product quality, but it can also obscure the origin of the catch, and may or may not be legal, depending on local regulations (The Pew Charitable Trusts 2018). Thus transshipment can be problematic from a regulatory, business, or consumer perspective because it decreases transparency; it may also facilitate human-rights abuses, and has been implicated in other crimes such as weapon and drug trafficking (Telesetsky 2015, Ewell et al. 2017). The situation is further complicated by the fact that transshipment often occurs in regions of unclear jurisdiction where policymakers and enforcement agencies may be slow to act against a challenge they cannot see.

Transshipment is also thought to be a factor in enabling illegal, unreported and unregulated (IUU) fishing, which is a global problem, extracting an estimated 11 – 26 million metric tons from the oceans each year (Agnew et al. 2009, Pauly & Zeller 2016). In addition to incurring an annual revenue loss of US$10 - 23.5 billion for legal fisheries, IUU fishing undermines fisheries management and conservation efforts and contributes to global overfishing (Agnew et al. 2009). It has been estimated that about a quarter to a third of all wild-caught seafood imports into major markets such as the USA and Japan could have been caught illegally (Pramod et al. 2014, 2017). Vessels transshipping part of their catch at sea or the mixing of catches from several fishing vessels from different regions, can obscure the traceability of seafood through the supply chain and introduce IUU catch into the global market under false labelling. The United Nation’s Food and Agriculture Organization (FAO) acknowledged this possible link between transshipment and IUU and developed guidelines and procedures for transshipment at sea to minimize illegal activities (FAO 2011). In addition, FAO launched an international plan of action to prevent, deter and eliminate IUU fishing, calling on flag states to improve monitoring and control of transshipments or to prohibit it entirely (FAO 2002). To date, transshipment is individually regulated by coastal and flag states and by Regional Fisheries Management Organizations (RFMOs). Some RFMOs, especially concerned about the laundering of high value-species such as tuna, restrict transshipment to port only (ICCAT 2006), prohibit certain fishing vessels from transshipping, or require onboard observers to be present (WCPFC 2009).

With increasing global demand for better seafood supply chain transparency and traceability transshipment has become an important, but yet poorly quantified focal point in the international trade of seafood. This can be addressed and resolved if each transshipment event is monitored and documented appropriately. New tools have emerged lately with the application of machine-learning technology to analyze vessel tracks based on Automatic Identification System (AIS) data, tracking the behavior of fishing vessels at a global scale and even in remote waters (McCauley et al. 2016, Kroodsma et al. 2018). Recently these methods have been expanded by researchers at Global Fishing Watch to analyze the behavior of reefers making it possible to detect and monitor transshipment at sea (Kroodsma et al. 2017, Miller et al. 2018).

Here we build and extend on this method to map and better understand the extent, spatial distribution, and role of transshipment for different fleets, gear types, and supply chains at a global scale. Using AIS data we ask where and when transshipment occurs, which fisheries and fleets are most involved in this practice and what proportion of high-seas catch is transshipped versus landed directly. We also apply this methodology to trace detailed seafood supply chains for tuna fisheries in the Indo-Pacific.

Materials and Methods

Likely encounters and fishing effort

Likely transshipment events ('encounters') were detected using satellite and tower-based AIS data between 2012 – 2017 as described by (Kroodsma et al. 2017). AIS was designed as a tool of maritime safety to avoid ship collisions. Transponders installed aboard vessels send position and vessel identification messages to receivers on other ships, land and satellites every few seconds. These messages can be used to reconstruct vessel tracks with high precision and allowed us to analyze their activity based on an automated analysis of movement patterns.

Likely encounters were identified by Global Fishing Watch as locations where two vessels which remained within 500 meters of each other for longer than 2 hours, traveling at less than 2 knots while at least 10 km from an anchorage (including ports). These parameters balance the need to detect vessel pairs in close proximity while recognizing our ability to identify long periods in which vessels are in immediate contact is limited by satellite coverage and inconsistent AIS transmission rates. Some vessels are known to transship within ports, but these events are more likely to be subject to surveillance and therefore we have focused on events that do not occur within the vicinity of port and the accompanying oversight. In this study we use a subset of the data analyzed by Miller et al. (2018), only including encounters where AIS data is available for both the reefer and the fishing vessel engaged in the encounter.

To exclude vessel meetings that occur within port, encounters were filtered so as to be more than 10 km from an anchorage (defined as docking in port or anchoring close by), by utilizing a global anchorage dataset developed by Global Fishing Watch and made publicly available at http://globalfishingwatch.io/anchorages.html. Briefly, the anchorages dataset was developed by applying an approximately 0.5 km grid to the globe using S2 grid cells (level 14) (http://s2geometry.io/). Using AIS messages from 2012 – 2016 from all vessel types, those grid cells where at least 20 vessels remained stationary for at least 48h where identified. For each grid cell, the mean location of the stationary periods was calculated, and this point was labeled as an anchorage. This method identified 102,974 anchorages and the mean location of an encounter was required to be at least 10 km from any anchorage.

A maximum encounter duration of 3 days was chosen to exclude encounters that significantly exceed expected catch offload durations. Such events likely represent vessels meeting for other reasons, such as repairs. This upper bound resulted in the removal of 97 events, representing less than one percent of the identified encounters.

Fishing vessels, refrigerated cargo vessels, fish carriers, and fish tender vessels were identified using vessel lists from the International Telecommunications Union (ITU) and major RFMO fleet registries. Additional vessels were identified by a vessel classification neural network developed by Global Fishing Watch to predict vessel types based on movement patterns. Vessels that were identified as likely reefers by this neural network were manually reviewed through web searches and national as well as RFMO registries. We do not expect that this list includes all vessels capable of receiving catch at sea, but it likely includes a majority of the large specialized reefers that transport fishing for much of the offshore fishing fleet. Of the 641 refrigerated vessels identified in this manner (Kroodsma et al. 2017), 501 were involved in likely transshipment events with AIS-tracked fishing vessels.

Fishing vessels included in this study were cross-checked for gear types through web searches using fleet registries and other reliable sources such as fishing company websites. To estimate the amount of catch landed directly by a fishing vessel versus catch brought to port via a reefer, we identified encounters and port/anchorage visits longer than 24h for each fishing vessel. For this analysis a vessel was not considered to have 'visited' a port or anchorage if it did not remain for longer than 24h to avoid assigning fishing effort to a port where a vessel was not present long enough to offload significant catch. For reefers, we identified the port visited following an encounter and the hours of fishing per fishing vessel that took place between events (the hours of fishing since the previous encounter or port visit). The fishing that preceded a port visit was assumed to have been landed in that port. Fishing hours that preceded an encounter were assumed to have been transferred from the fishing vessel to the reefer and offloaded in the next port that the reefer visited. The total fishing hours were aggregated by gear and accordingly attributed to ports (Russia considered separately from Asia and Europe).

Fishing activity and vessel gear type were classified following the methods described by Kroodsma et al. (2018). Briefly, two convolutional neural networks are trained on data from fleet registries, logbooks and data labelled by experts, to identify vessel types and classify their behavior (transiting, fishing) based on movement characteristics as seen in the AIS data.

Tuna supply chain

Data on supply chains for three reefers and 16 fishing vessels transshipping catch at sea was supplied by industry and consisted of official transshipment documentation as well as Captain's Statements. Based on the vessel identification numbers and details on date, location and vessels involved in the transshipment given, AIS tracks were reconstructed for the three reefers and 13 of the 16 fishing vessels from raw AIS data supplied by Global Fishing Watch. Industry-recorded encounters were compared against the AIS-based detection method for transshipments as described above.

Results

Likely transshipment events (fishing vessel-reefer encounters at sea detected from AIS positions of vessels within 500 meters of each other and lasting longer than 2 hours, traveling at less than 2 knots while at least 10 km from shore, hereafter called 'encounters'), were identified from 22 billion individual AIS position signals where AIS data were available for both reefer and fishing vessel engaged in the encounter (Fig. 5.1). AIS messages provide detailed information on vessel identity and behavior, and have become more widely available since 2012 (McCauley et al. 2016, Kroodsma et al. 2018). Novel machine learning algorithms allowed us to automatically detect and map encounters between fishing and refrigerated cargo vessels at sea. Using a subset of the global database developed by Global Fishing Watch (Kroodsma et al. 2017) including AIS tracks from both reefers and fishing vessels, we quantified the spatial distribution of encounters between fishing vessels (focusing on four major gear types) and refrigerated cargo vessels and estimated the fishing effort (in hours spent fishing) as a proxy for the catch that was accumulated between encounters or port calls (see methods below for more detail). Between 2012 and the end of 2017 we observed 501 reefers meeting up with 1,856 fishing vessels in 10,510 likely transshipment events worldwide. The refrigerated cargo vessels involved comprise a variety of types, including fish carriers, fish processors, and a small number of fish tenders.

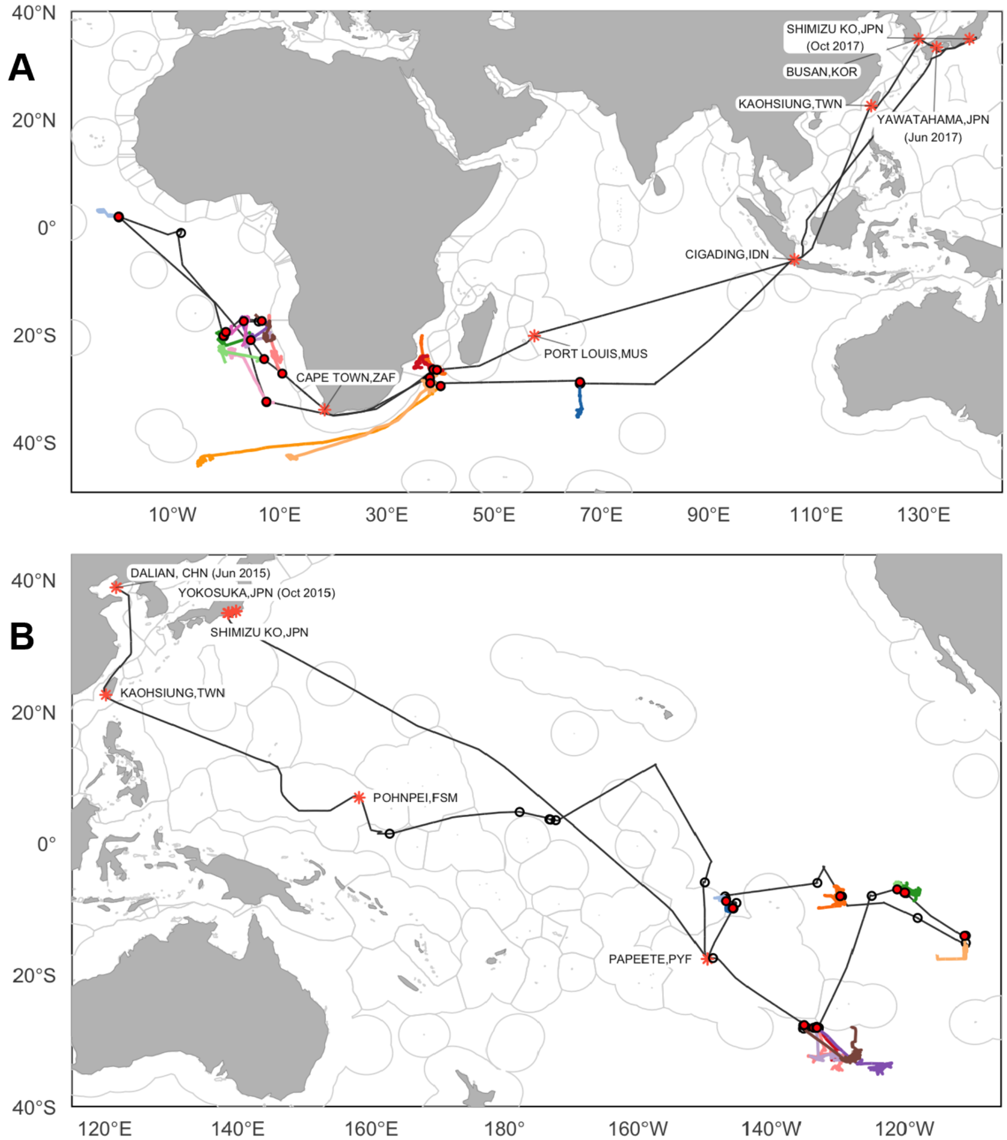

Fig. 5.1 Examples of transshipment of catch at sea. Shown are example AIS tracks of reefer (black) and fishing vessels (colors), port calls (asterisk), likely transshipment encounters (red circles) and potential encounters (white circle) in the Atlantic (**A**) and Pacific (**B**). Exclusive economic zones are outlined in light grey. Note that tracking data for fishing vessels are missing for some likely encounters but reefers exhibited behavior consistent to an encounter.

Taken together, 35% of all observed transshipment encounters occurred on the high seas, while 65% took place within Exclusive Economic Zones (EEZs) where most global fishing occurs (Kroodsma et al. 2018). A large fraction (39%) of all detected encounters occurred in the Russian EEZ, with the remainder (61%) spread over 41 other nations' EEZs. Excluding Russia, 57% of likely encounters took place on the high seas.

Fishing vessels engaged in transshipping were mostly trawlers (53%) and longliners (21%), the former being more active in shallow continental shelf waters, the latter concentrating on the high seas. Squid jiggers (13%), fishing vessels using pots and traps (7%), and purse seiners (1.2%) contributed less to global transshipment events detected from AIS data.

Transshipping from trawlers was most common in EEZs in the northern hemisphere, most notably in Russian waters, whereas the majority of transshipments from longliners, purse seiners and squid jiggers occurred on the high seas, with hot spots off West Africa, in the South Indian Ocean and the equatorial Pacific (Fig. 5.2).

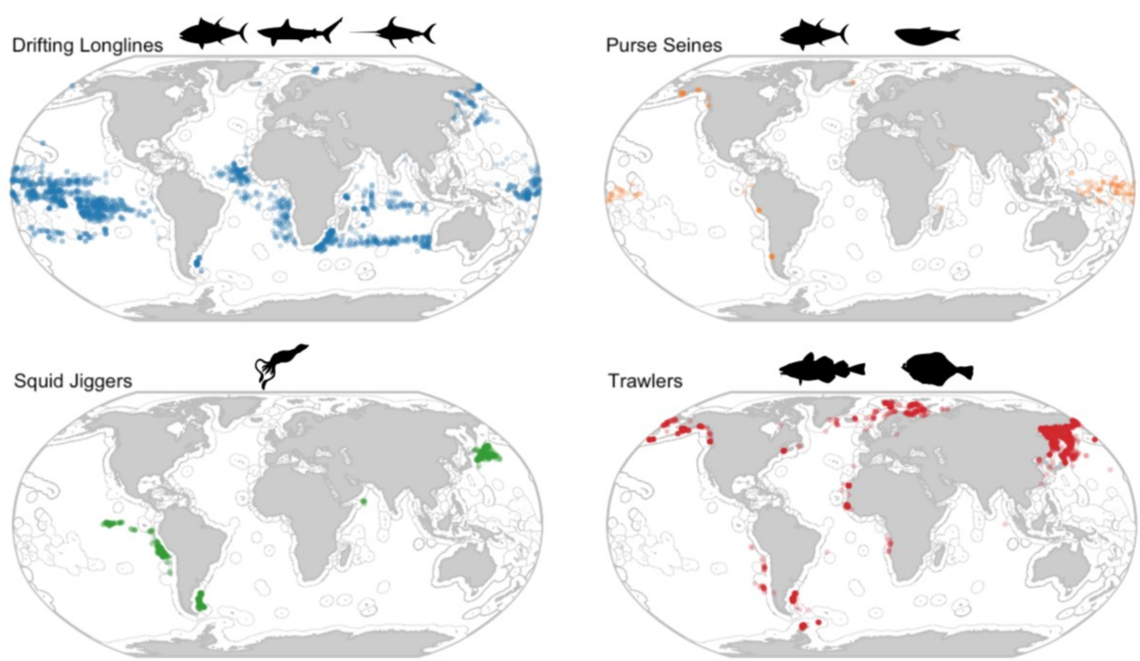

Fig. 5.2 Global patterns of transshipment for different fishing gears. Shown are all likely encounters (colored dots) between reefers and fishing vessels as identified from AIS data spanning 2012 to 2017 and separated by fishing gear type. Exclusive Economic Zones are outlined in light grey, pictograms illustrate major target species.

The average duration of likely transshipment events identified in the AIS data was 11.6 hours (median 7.3 hours) which is close to the 9.5 hours reported in transshipment documentation (see below). Fishing vessels transshipped their catch to a reefer roughly once a month. Most reefers traveled to meet the fishing vessels at or close to the fishing grounds (Fig. 5.1) whereas fishing vessels only traveled relatively short distances (mean distance 122 km, median distance 42 km) to meet a reefer.

For the majority of time vessels spent fishing before meeting a reefer they were located in EEZs (Fig. 5.3 A and B). Catch from more than three quarters of all observed fishing in EEZs (86%) was landed directly, whereas only 14% was transshipped. Transshipment was much more prevalent on the high seas, with nearly half (45%) of catch from observed fishing effort on the high seas being transshipped (Fig. 5.3). In EEZs, trawlers predominated landings and transshipment events, whereas on the high seas longline fishing dominated both in terms of landed and transshipped catch, followed by squid jiggers (Table 5.1 and Fig. 5.4). Trawlers predominantly fished and transshipped in Northern hemisphere temperate waters, whereas longliners operated globally in tropical and subtropical waters, and squid jiggers were observed in international waters along the EEZs of South American countries both in the Pacific and Atlantic (Fig. 5.2).

Table 5.1 Direct landing or transshipment of catch in Exclusive Economic Zones versus the high seas. Shown are the percentage of fishing hours landed directly in port by fishing vessel or transshipped at sea and landed by reefer. Data are separated by fishing gear and for Exclusive Economic Zones (EEZ) and the high seas (HS, shaded grey). Percentages are given for fishing in all EEZs and for the Russian EEZ separately due to outstanding importance of transshipment for Russian fleets.

	IN EEZ				**IN HS**	
	landed directly	*landed directly from Russian EEZ*	*transshipped*	*transshipped from Russian EEZ*	*landed directly*	*transshipped*
Trawler	84.3	97.9	81.2	97.2	41.8	15.3
Longliner	8.1	1.2	13.7	1.8	47.0	64.5
Purse seiner	7.1	0.5	0.6	0.06	8.3	0.1
Squid jigger	0.5	0.4	4.4	0.9	2.9	20.1

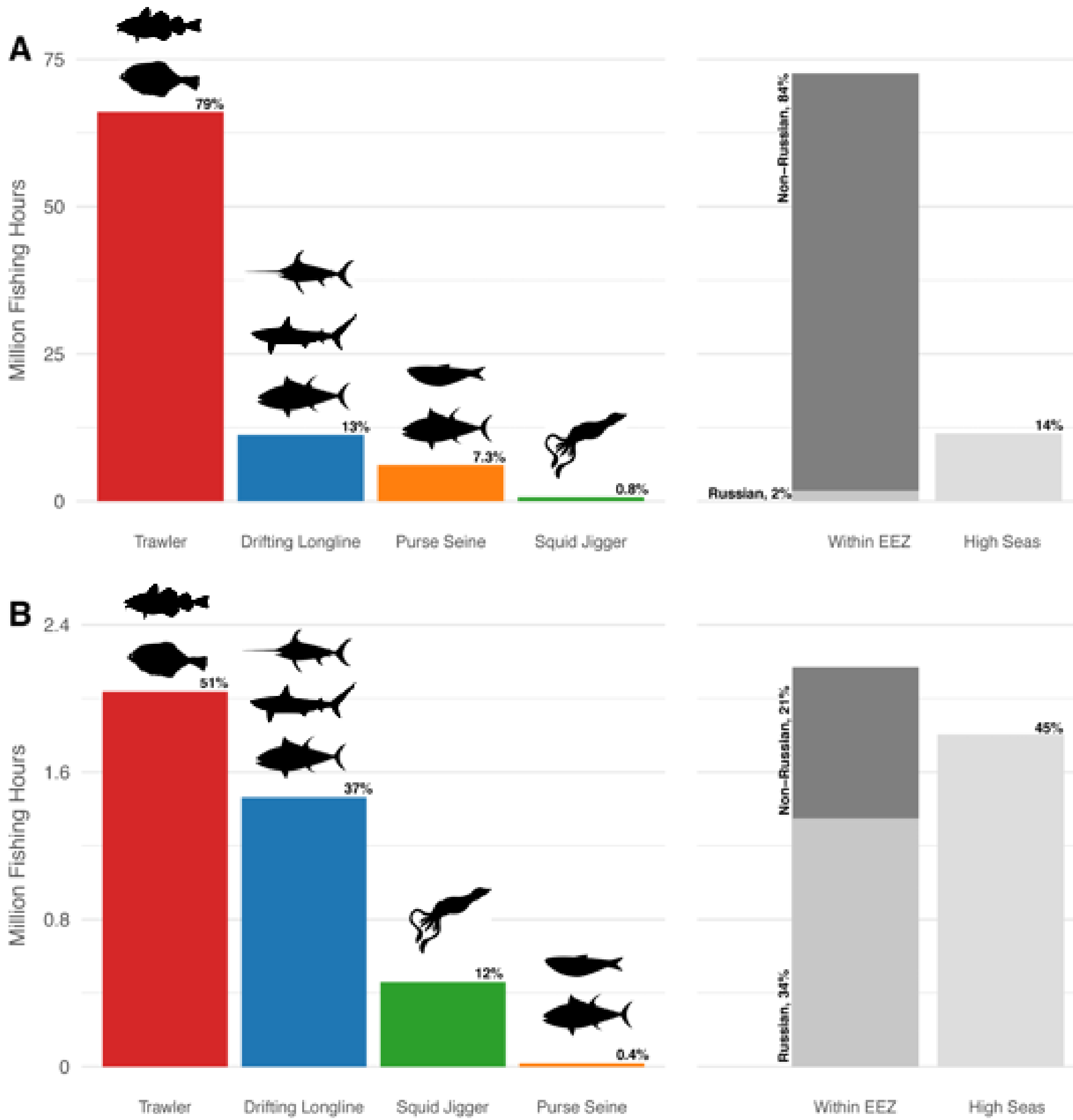

Fig. 5.3 Relative extent of transshipment for different fishing gears. Shown is the fishing effort (estimated fishing hours), that is landed directly in port (**A**) versus transshipped and brought to port by reefer (**B**). Data are separated by fishing gear type (left) and for Exclusive Economic Zones (EEZ) versus the high seas. Data includes fishing vessels that at least once have met up with a reefer. Gears represent more common gears used by fishing vessels involved in encounters. Pictograms denote major target species by gear type.

A fishing vessel's voyage may be broken into three segment types of varying durations. For short daily fishing trips, the entire voyage might be characterized by the segment of time between two anchorages (including both docking in port or anchoring nearby). Longer trips, which include likely transshipment encounters, can be divided into additional segments such as the time between an anchorage and an encounter at sea, or the time between two sequential encounters. Excluding the upper and lower 5% of the data to eliminate implausible outliers caused by data gaps (Fig. C.1), we found that fishing vessels that undertook voyages characterized solely by an anchorage exit and a return (no transshipment involved) spent about 18 days at sea (median 6 days) and fished about 46 hours (median 23.5 hours). This estimate is influenced by short coastal fishing trips with vessels returning to port every day. For fishing vessels engaging in transshipment, we found the time between an anchorage exit and a fishing vessel's first likely transshipment encounter was about 50 days (median 37 days) during which time the vessel fished for an average of 100 hours (median 74 hours). Between transshipment encounters, we found fishing vessels met with a reefer about every 31 days (median 19.5 days) and fish about 132 hours (median 135.5 hours). The longer time between anchorages and first transshipment encounters is likely due to the time fishing vessels spent traveling to their fishing grounds and the fact that some encounters are not identified due to missing AIS signals (lack of satellite coverage and/or switching off of AIS transponder).

Out of 33 flag states observed to operate reefers, Russia accounted for almost a third (32%), followed by Panama (20%) and Liberia (7%), the latter two representing so-called flags of convenience (FoCs), flags of states characterized by loose regulation and limited oversight (Fig. C.2 A). About 41% of all reefers were flagged to FoCs, or 60% when excluding Russia. Fishing vessels from 47 nations were found to encounter those reefers and engage in likely transshipment; again, a majority from Russia (26%), followed by China (20%) and Taiwan (15%) (Fig. C.2 B). Encounters of fishing vessels with reefers flying FoCs were more prevalent on the high seas than in EEZs for all gear types, especially for squid jiggers (78% of all high seas encounters compared to 27% within EEZs) and longliners (62% to 25%, respectively).

Testing for a correlation between the number of likely transshipment encounters and regional extent of IUU as estimated by FAO area (Agnew et al. 2009), we found a weak positive, but non-significant relationship (p = 0.1626) (Fig. C.3). FAO area 61 (Northwest Pacific), emerged as a notable outlier of this analysis, with both a high percentage of IUU (33%) and by far the highest number of likely transshipment events (44% of total).

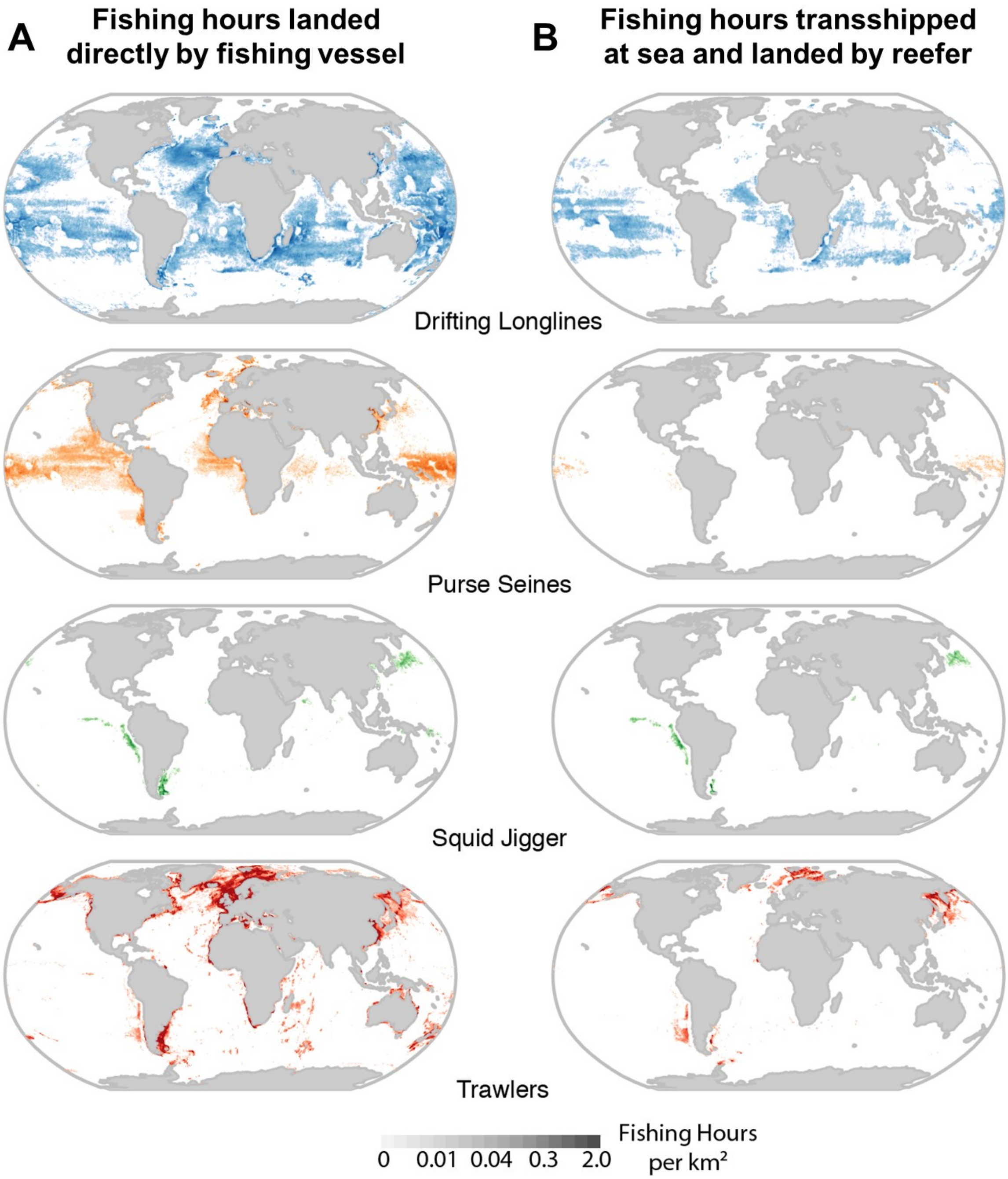

Fig. 5.4 Spatial patterns of landed versus transshipped fishing effort. Shown is the spatial distribution and intensity (fishing hours per km-2) of fishing effort for each gear type landed directly by fishing vessel (**A**) or by reefer after transshipment at sea (**B**) between 2012 to 2017.

Based on information provided from a tuna processor and retailer we were able to reconstruct detailed supply chains for tuna transshipped to and landed by three reefers flagged to China, Taiwan and Panama and operating in two of the global hot spots we identified here: the south Indian Ocean and the equatorial Pacific (Fig. 5.5). These three vessels spent an average of eight days (1 to 23 days) in port and about 50 days at sea (23 to 96 days, excluding short transits from port to port) and received an average amount of 57,500 kg of catch (mostly albacore tuna [*Thunnus alalunga*]) per transshipment from 16 fishing vessels flagged to either China or Taiwan. Of these fishing vessels, AIS data were available for 13 (Fig. 5.5). Using the transshipment location as noted in the reefer's documentation, we were able to match seven of the 13 documented transshipment events to the AIS data used in this paper. For six events it was not possible to identify a likely transshipment event (within 100 km radius) from the AIS data.

Based on AIS tracks and industry documentation, we estimate that tracked tuna fishing vessels fished for about two to three weeks before meeting with a reefer to offload their catch. The reefer returned to port to land the transshipped catch about once a month, depending on the distance from port and the number of fishing vessels encountered. In processing facilities in or close to the port of landing the whole fish was processed into loins and shipped in sealed containers to canning facilities, in this case located in the USA. This takes four to eight weeks, depending on location of the port. Re-processing and canning happens over another four weeks with a subsequent distribution to retail within two to twelve weeks. It thus takes about half a year on average (18 to 35 weeks) from the catch of albacore tuna to the canned final product on the shelf. Along the entire supply chain, the fish have traveled an average 17,000 km (13,000 to 20,000 km, excluding traveling on the fishing boat and transport to final retail) with about five discrete steps involved, including post-production steps such as shipment of cans (Fig. 5.5).

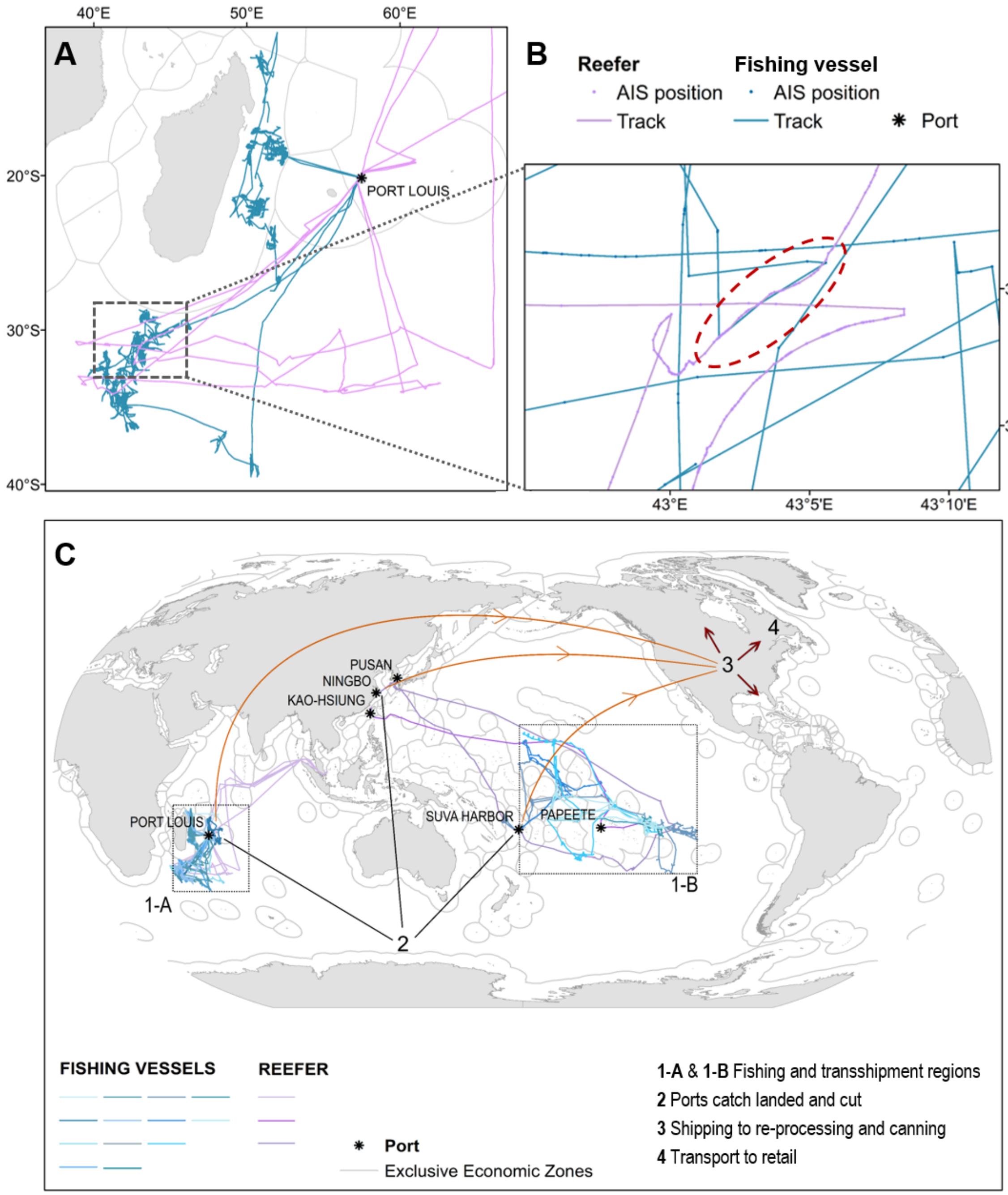

Fig. 5.5 Case study of transshipment of tuna. Shown is the path of albacore tuna from fishing location to retail shelf. Reefer (purple) and fishing vessel (blue) tracks, area of fishing and transshipment denoted by dashed black rectangle, Exclusive Economic Zone boundaries in light grey. **A**: Fishing and transshipment off Mauritius, port call into Port Louis, **B**: close-up of transshipment event (dashed red circle). **C**: Tracks of three reefers and 13 fishing vessels from January 2017 to February 2018. (**1-A**) and (**1-B**) (dashed rectangles) denote fishing and transshipment areas, (**2**) ports (asterisk) where reefers landed whole fish and fish is cut, (**3**) transport to re-processing and canning facilities, and (**4**) transport of final product to retail.

Discussion

In the last decade, transshipment of catch at sea has become a focal point in the international discussion surrounding seafood supply chain transparency, especially for fisheries operating in distant waters and featuring complex supply chains. Fish commonly passes from producers (individuals/companies operating fishing vessels) to fish brokers, who aggregate catches upon landing or transshipment to a reefer and arrange for sale to processors and distributors. Unsurprisingly, traceability of products becomes more complicated with increasing supply chain length, complexity and levels of aggregation of catch. While fish landed directly in port by fishing vessels is usually documented by vessel before aggregation of catches from multiple sources, this documentation is less precise for catches transshipped at sea.

Here we build on a global database of transshipment encounters developed by Global Fishing Watch (Kroodsma et al. 2017, Miller et al. 2018), mapping empirical observations of transshipment at sea by gear and region and connecting it to supply chains to highlight the role, scale and importance of transshipment in the global seafood trade. We found that while transshipment is occurring in all oceans and across 42 EEZs (Miller et al. 2018), it is more common in distinct hot spot areas on the high seas (e.g. south Indian Ocean, equatorial Pacific), in some EEZs (e.g. off Russia, west Africa), for some gear types (trawlers, longliners) and involving few dominant states that flag a majority of reefers (Russia, Panama, Liberia).

Transshipment is mostly seen close to fishing grounds (Fig. 5.2) as it is common practice for fish traders to arrange for the reefer to meet the fishing vessels. The distribution of transshipment activity and the types of fishing vessels transshipping catch depend on the nature, value and volume of target species and can be useful indicators for fisheries managers to pinpoint areas and fisheries where monitoring and documentation should be enhanced.

Observed transshipment events within EEZs largely involved trawlers, likely fishing on the continental shelves for demersal or coastal-pelagic species. As these fisheries generate high-volume catches, transshipment enables vessels with limited hold capacities to continue fishing. On the high seas, more than half (excluding Russia) likely transshipment events involved longline fishing vessels, presumably transshipping highly migratory species such as tuna, sharks and billfishes (swordfish, marlins) (Gianni & Simpson 2006, Cullis-Suzuki & Pauly 2010). Few longline vessels have adequate deep-freezing facilities, thus quick transshipment to reefers is essential to maintain high quality and market prices (The Pew Charitable Trusts 2018). This suggests that the type of catch (high volume or high value) and its location shape the infrastructure of the supply chain involved and thus can be an indicator which fisheries and supply chains might be most susceptible to illicit activities surrounding transshipment, thus warranting closer monitoring, control and surveillance.

Some fishing fleets rely heavily on the use of reefers regardless of the type of fishing. More than a third of all observed transshipments were conducted between Russian-flagged reefers and fishing vessels in the Russian EEZ and the Bering Sea, areas with poor monitoring of transshipment (Ewell et al. 2017) and a history of illegal fishing. Russia's fishing fleet largely dates back to the Union of Soviet Socialist Republics and, struggling to meet targets set to close a gap in food supply after World War II, Soviet fishing fleets were restructured in the 1950s and 1960s to increase operation time and range (Sealy 1974, FAO 2007). Fishing operations were centered around mother ships and fish carriers to supply the fishing fleet and process their catch (Sealy 1974); these historical developments may partly explain the importance of transshipment and the central role of reefers in Russian fisheries today (Miller et al. 2018). In addition, a strong link to the nearby Chinese market (57% of all fish imports to China come from Russia) further favors transshipment in Russian waters and under Russian flag (Pramod et al. 2014). Relatively poor monitoring, low compliance, weak enforcement and high levels of transshipment enables IUU fishing for Russian pollock, crab and salmon, which are imported to the USA and Europe following reprocessing in China (Pramod et al. 2014). These fisheries are contributing to high estimated prevalence of IUU (33%) in the Northwest Pacific (FAO area 61) (Agnew et al. 2009, Pramod et al. 2014) (Fig. C.3). However, the overall correlation between AIS-detected transshipment and estimated IUU fishing is weak (Fig. C.3), possibly owing to large uncertainties in quantifying both processes, and a scale mismatch between localized transshipment observations and FAO-area IUU estimates. For improved analysis, more regional knowledge on IUU fishing is required.

No comprehensive global regulations or codes of conduct for transshipment exist. Next to regulations by RFMOs for their convention areas (see below), it is up to individual states to regulate transshipment within their own EEZ and for vessels flying their flag. Following FAO recommendations (FAO 2002), some nations, such as Thailand, Nauru and Indonesia, have temporarily or permanently banned transshipment in their waters or for vessels flying their flags (Ewell et al. 2017). Some flags feature weaker regulations and enforcement and less oversight, particularly so-called flags of convenience (FoCs) (following definition by Miller & Sumaila (2014)). The high prevalence of FoC-flagged reefers found in this study (41% of total observed, 60% if excluding Russia) and the fact that they primarily engage in transshipments in areas beyond national jurisdiction might compromise transparent documentation of seafood supply chains and warrants further consideration.

In the international waters of the high seas responsibility for fisheries management lies with the RFMOs. While some RFMOs have developed measures to document and regulate transshipment such as required onboard observers and an electronic Vessel Monitoring System (VMS) (McCauley et al. 2016), this is not globally coordinated (Ewell et al. 2017). A recent study found that out of the 17 RFMOs active on the high seas five have mandated a partial and just one a total ban of transshipment at sea (Ewell et al. 2017). Thirteen RFMOs mandate some form of vessel tracking in relation to transshipment such as VMS and ten require an onboard observer. For example, the Western and Central Pacific Fisheries Commission (WCPFC) requires observer coverage and a notice of planned transshipments at least 36 hours prior (WCPFC 2009) while the Indian Ocean Tuna Commission (IOTC) allows transshipments from large tuna longliners only (IOTC 2014). Fishing vessels using certain gear types such as purse seines are prohibited to tranship in some areas, which is likely one reason why only 1.2% of all fishing vessels involved in encounters seen in this study are purse seiners.

How these mandates and regulations are enforced on the water, however, remains questionable and documentation by authorities is hard to access. For instance, more than 100 likely encounters between fishing vessels and reefers were observed between 2012 and 2017 in the convention area of the South East Atlantic Fisheries Organization (SEAFO) where all transshipment of fishery resources covered by the Convention is banned (SEAFO 2016) (Fig. 5.1). One such instance involving a likely encounter between a Japanese longline vessel and a Liberian reefer is highlighted in Fig. C.4. It remains unclear if the likely encounters observed within the convention area are transshipping fish from resources covered by the SEAFO convention, resources covered by another convention with overlapping area (in this case the International Commission for the Conservation of Atlantic Tunas [ICCAT] and the Commission for the Conservation of Southern Bluefin Tuna [CCSBT], both regulating tuna and tuna-like species), or if the encounter constitutes a mere resupplying of the fishing vessel by the reefer (which, however, appears not to be exempt from the term 'transshipment' by SEAFO). This highlights the importance of proper monitoring and transparent documentation of all encounters at sea, whether they are to transship catch or to resupply.

Monitoring of remote waters and the high seas can be facilitated through the use of AIS data, complementing existing monitoring, control and surveillance tools (Dunn et al. 2018). This combination of various tools is useful to create a complete picture of global fisheries and seafood supply chains. Looking at tuna fisheries in two global hot spot areas (south Indian and equatorial Pacific Ocean, see below) and tracking known transshipment events using AIS data we found that only seven out of 13 (or 54%) documented transshipment events could be reconstructed using AIS data. This is likely due to a combination of gaps in the AIS data as well as poorly recorded transshipment locations. Hence our estimates of the global prevalence of transshipment should be seen as very conservative; the true extent is evidently much higher.

As discussed in detail elsewhere (Natale et al. 2015, de Souza et al. 2016, Kroodsma et al. 2018) some important caveats and limitations apply to the use of AIS data in general: while coverage by AIS-capable satellites is continuously increasing, some areas may not be covered 100% of the time and transshipment events in these areas might go unnoticed some of the time. Furthermore, AIS transponders onboard vessels can be manually switched off or location data can be manipulated (Kroodsma et al. 2018). For the detection and subsequent classification of a likely transshipment event in this study, AIS data of both the reefer and the fishing vessel need to be available and correspond to the chosen characteristics of an encounter. Where no AIS data for fishing vessels involved in encounters is available, 'loitering' behavior of the reefer may still be indicative of likely transshipment events (Miller et al. 2018). However, due to the missing AIS data for fishing vessels involved in those events we excluded these from our data. This reduces the numbers of encounters analyzed and may bias results towards transshipment events including large, AIS-equipped vessels operating offshore. Global patterns of other potential transshipments events though are largely similar to those shown here, and discussed by Miller et al. (2018). Lastly, gaps in the AIS data might also influence the calculations of fishing hours landed versus transshipped. If an encounter or port call is not included due to missing data, fishing hours might be overestimated or wrongly allocated to the following transshipment or encounter.

Based on a fully documented industry supply chain we illustrated the voyage of albacore tuna from the hook to a retailer's shelf. In this case, individual fish travel roughly 17,000 km after catch, over a time span of up to half a year, changing boats, owners, and processing facilities several times (Fig. 5.5). Ideally, every step of this complex supply chain is documented and recorded electronically, at sea and in port, and the documentation we received from industry illustrates how this can be done. At-sea documentation includes fishing location, gear used, and amounts caught by species (ideally also recording bycatch), time, date and location of all transshipment events during that trip as well as identity of vessels involved, catch already transported by the reefer, and all ports visited. Some of this information was not included in the transshipment documentation used in this study: fishing locations were recorded only by RFMO or ocean area and overall information on the origin of all catches transshipped by reefers servicing fishing vessels for more than one buyer appears to be generally not available.

The entry of fish to the market via port is a key point in supply chains to require and verify documentation and preclude IUU catch from landing, as included in the recent Port State Measures (Swan 2016). On land, further documentation includes method of delivery (fishing vessel direct, by reefer, containerized via another port) and production code or lot numbers specific to the fishing vessel trip the fish was caught. Following landing, catches ideally are binned in sealed containers corresponding to these codes and lot numbers, which are carried though all levels of processing to maintain traceability of the fish to the final product.

As we presented here, AIS data enables independent verification of vessel activities, including transshipment (McCauley et al. 2016), expanding and complementing existing monitoring and documentation tools. Ultimately, improved legislation and transboundary management may want to include mandatory AIS to ensure increased traceability and transparency in supply chains (Dunn et al. 2018, The Pew Charitable Trusts 2018).

Conclusion

In this analysis, we highlighted global hot spots of transshipments such as the Russian EEZ and the high seas, especially off West Africa, in the Southern Indian Ocean and (most prominently) the tropical Pacific where high-value species such as tuna are fished. Trawlers in territorial waters and longliners on the high seas contributed a large majority of likely transshipment events. To reduce the probable introduction of IUU catch into the supply chain, strict monitoring and documentation of each transshipment event is needed, especially if it takes place in international waters. AIS data are ideally suited for long-range monitoring and surveillance of vessel movements and new methods are available to independently detect and document likely transshipment events in addition to documentation provided by vessels and observers. Therefore, AIS-based monitoring of transshipment, coupled with improved regulation and oversight, holds promise for improving fisheries management and trade practices on the high seas, and elsewhere.

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Author Contributions

Concept and design by KB and NM. Data preprocessing done by NM, data analyses by KB and NM and discussion and writing by KB, NM and BW. Data and materials available through Global Fishing Watch and upon request to research@globalfishingwatch.org

Appendix C

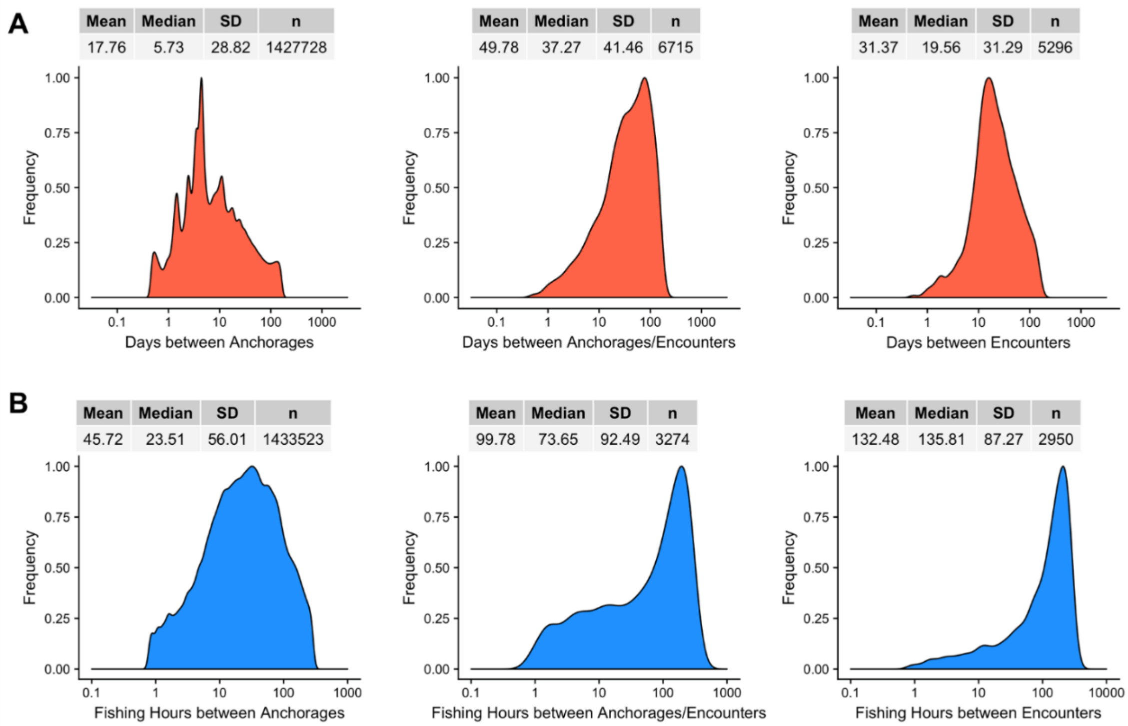

Fig. C.1 Frequency distribution of days at sea (A) and hours spent fishing (B) between two anchorages, anchorage and encounter, and between two encounters. Note that the upper and lower 5% of data have been excluded to avoid extreme outliers likely caused by missing AIS data.

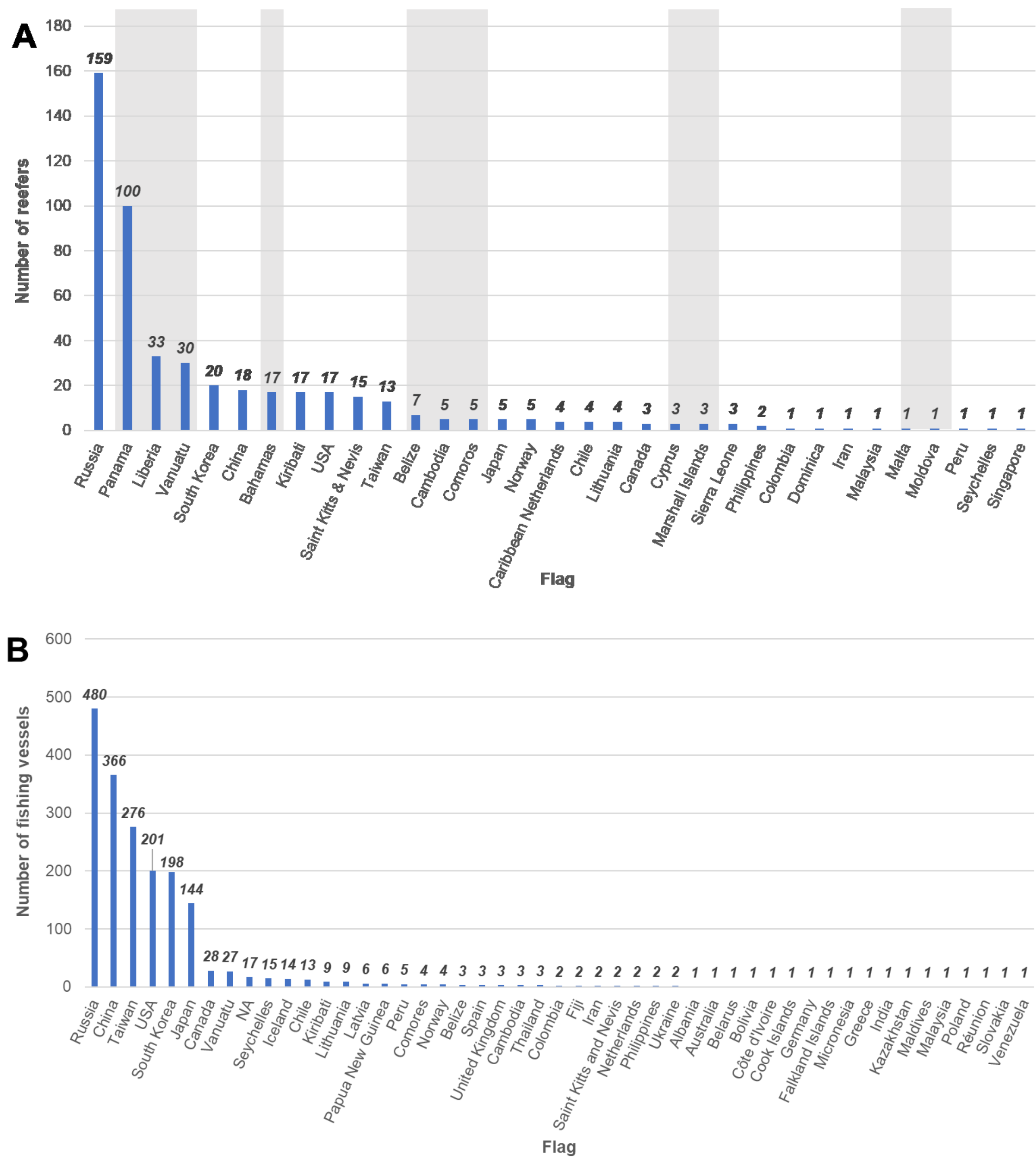

Fig. C.2 Number of reefers (A) and fishing vessels (B) involved in likely encounters between 2012 and 2017 worldwide by flag. Flags of convenience for reefers marked in grey.

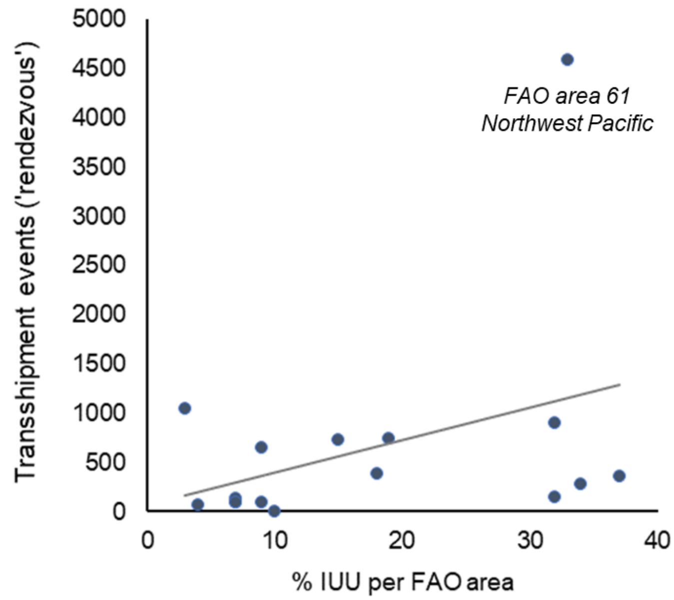

Fig. C.3 Correlation between the number of rendezvous from 2012 to 2017 and illegal, unreported and unregulated (IUU) fishing by FAO region as reported by Agnew et al. 2009 (p = 0.1626).

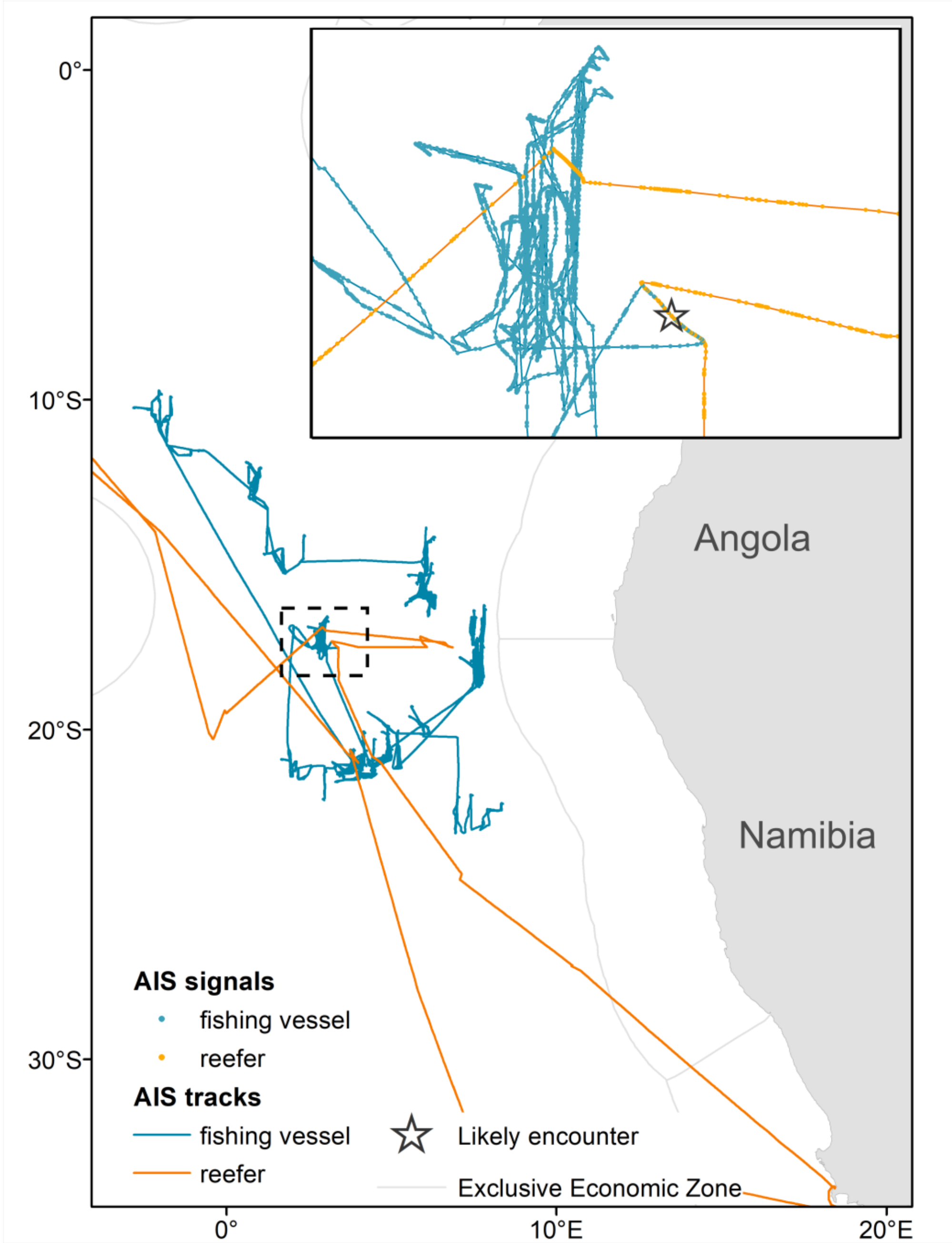

Fig. C.4 Likely encounter between reefer flagged to Liberia (orange) and a Japanese longline fishing vessel (blue) off the west coast of Southern Africa. Insert (dashed line): close-up of likely encounter (star). Dots indicate AIS position messages.

CHAPTER 6
NOT ALL WHO WANDER ARE LOST: SPATIAL PROTECTION FOR LARGE PELAGIC FISHES

Abstract

Spatial protection measures are becoming increasingly ubiquitous in fisheries management and marine conservation. Implemented for a variety of objectives from stock rebuilding to biodiversity protection and ecosystem management, spatial measures encompass temporary fisheries closures as well as established marine protected areas with varying levels of protection. Ecological and economic benefits from spatial closures have been demonstrated for many reef and demersal species, but remain debated and understudied for highly migratory fishes, such as tunas, billfishes, and pelagic sharks. Here we review the spatial extent of fisheries closures and protected areas, which collectively cover close to 15% of global ocean area. We synthesize results from modelling and tagging studies as well as fisheries-dependent research to provide an overview of the efficacy and benefits of spatial protection for large pelagic fishes as well as their associated fisheries. While species life history attributes differ widely, species that migrate along known routes, aggregate around oceanographic features, or show homing behaviour to predictable areas are more likely to benefit from spatial protection tailored to their biology and life history. Reviewing effects of existing spatial closures for large pelagics at national and international level, we find that a combination of fisheries management and spatial protection measures appears most effective to protect and rebuild highly migratory fish stocks. We suggest a tailoring of spatial protection to the biology of large pelagic fishes, including improved protection for aggregation sites and migration corridors as they currently appear to be important—yet overlooked—to safeguard overfished stocks and protect biodiversity.

Introduction

The last two decades have seen a notable increase of ocean area placed under some form of spatial protection (UNEP-WCMC & IUCN 2016). These areas include both large-scale fisheries closures aimed at safeguarding heavily fished stocks, and marine protected areas (MPAs) designed to protect marine biodiversity more broadly (IUCN 2018a) (Fig. 6.1 and Table D.1). The establishment of MPAs in particular has gained much attention, especially due to the creation of several very large MPAs (>100,000km^2) in recent years (McCauley et al. 2016, O'Leary et al. 2018). Although the conservation of biodiversity is the primary objective (IUCN 2018a), another common, often more implied, goal of MPAs is the delivery of benefits to nearby fisheries by increasing local fish abundance, biomass, and larval supply.

The ability of closed areas to meet conservation goals has been demonstrated numerous times (e.g. Kelly et al. 2000, McClanahan & Mangi 2000, Murawski et al. 2000, Roberts et al. 2001, Gell & Roberts 2003, Halpern 2003, Goñi et al. 2006, 2008, Lester et al. 2009, Follesa et al. 2011, Russ & Alcala 2011, Alemany et al. 2013, Kerwath et al. 2013, Costello 2014) and a number of studies have shown benefits of closed areas for nearby fisheries, mainly through increasing local stock biomass, protecting vulnerable life stages, and the spillover of eggs, larvae and adult fish from the protected area to nearby regions. Yet, most empirical examples of successful spillover focus on small, nearshore MPAs and non-migratory species such as lobster (Kelly et al. 2000, Follesa et al. 2011), clams (Tawake et al. 2001), scallop (Murawski et al. 2000) and reef fishes (McClanahan & Mangi 2000, Stobart et al. 2009, Da Silva et al. 2015, Friedlander et al. 2017, Tewfik et al. 2017). In contrast, potential benefits of closed areas for large pelagic fishes such as tunas, billfishes and sharks, have received less scientific attention, mostly because their highly migratory nature may present a substantial challenge for the design and implementation of effective spatial protection and thus, these approaches have traditionally been less common for protecting these species (Galland et al. 2016).

Many large pelagic fish are of high commercial value but long life spans and high age at maturity make them especially vulnerable to overexploitation (Collette et al. 2011). For example, many populations of large tuna species (i.e., Atlantic [*Thunnus thynnus*] and southern bluefin [*Thunnus maccoyii*], bigeye [*Thunnus obesus*], yellowfin [*Thunnus albacares*] tuna) have been depleted to 10-25% of their virgin spawning biomass (SSB_0) (Minte-Vera et al. 2014, IOTC 2017, CCSBT 2017, ICCAT 2017c) with extreme depletion (i.e., > 95%) observed in Pacific bluefin [*Thunnus orientalis*] (ISC 2016b). In addition to changes in abundance, the spatial ranges of all three bluefin tuna species have all shrunken significantly since 1960 (Worm & Tittensor 2011).

Similar changes have been observed in some billfish (swordfish, marlins and sailfish). Recent assessments suggest five of seven billfish stocks in the Atlantic Ocean are overfished (ICCAT 2017c). In the Pacific Ocean the stock of striped marlin has been subject to overfishing since 1977 (ISC 2015), whereas blue marlin (*Makaira mazara*) appears healthy (ISC 2016a). Of five assessed Indian Ocean billfish stocks, only swordfish (*Xiphias gladius*) is considered healthy (IOTC-WPB15 2017). Furthermore, global range contractions have been observed for black (*Istiompax indica*), striped (*Kajikia audax*), and white marlin (*Kajikia albida*), as well as sailfish (Worm & Tittensor 2011).

The situation appears particularly dire for sharks – more than half of pelagic sharks and rays are thought to be threatened with extinction (Dulvy et al. 2008, 2017). Like billfish, these species are susceptible to high mortality from incidental capture in tuna longline fisheries (Gilman et al. 2008, 2017), yet they are also targets of directed fishing to meet the demand of a lucrative Asian market for their fins (Clarke et al. 2012, Patterson et al. 2014). Given the high degree of unreported catch incurred by both practices, as well as noted incidents of illicit shark fishing in protected waters, much of the world's shark catch is contextualized as illegal, unreported, or unregulated (IUU) (Agnew et al. 2009, Galland et al. 2016, Ward-Paige & Worm 2017).

The 1982 United Nations Convention on the Law of the Sea (UNCLOS) endowed nation states with sovereignty in managing fish and other species within 200 nautical miles of their coasts (i.e. their exclusive economic zone, EEZ). While this arrangement suggests most marine populations can be managed domestically it presents a challenge for large pelagic fish whose distributions straddle many EEZs and the High Seas. To facilitate the cooperative management and long-term conservation of these species between the countries catching them, the 1995 UN Fish Stocks Agreement (UNFSA) resulted in the proliferation of regional fisheries management organizations (RFMOs)—of which there are now five specifically focused on large pelagics. Regional fisheries management organizations today are the primary institutions through which conservation and management measures (CMMs) for tuna, sharks and billfish are discussed, adopted and implemented.

However, despite the efforts of many RFMOs, total fishing pressure for many large marine predators has proven hard to control and monitor. This is complicated by differing biological characteristics of target species (McKechnie et al. 2016, Pons et al. 2017), competing fishing interests between national fleets (Bailey et al. 2010, 2013, Squires 2013), the unequal conservation burden between nations in the global North and South (Hanich & Ota 2013, Hanich et al. 2015), limited transparency in decision making (Polacheck 2012), and incomplete monitoring of fishing activity (Gilman 2011, FAO 2012, Dunn et al. 2018). Additional multi-lateral international agreements (e.g. *Convention on International Trade in Endangered Species of Wild Fauna and Flora [CITES], Convention on Migratory Species [CMS]*) have also sought to promote the conservation of pelagic fish, mostly through regulation of trade. However, to date, these agreements have had limited success for tunas (Webster 2011), although the listing of certain shark species has resulted in stricter measures around their retention at the RFMO level and likely motivated private-sector commitments (e.g., airlines, marine shipping companies) focused on reducing shark fin trade (Shea & To 2017).

Over the last decade a wide range of spatial protection measures for large pelagic fish have been put in place both unilaterally and through RFMO CMMs (Table D.1). These include (but are not limited to): Prohibition of purse seining in Pacific high seas pockets (Pala 2010), annual fisheries closures for specific gears of defined subsets of management areas (ICCAT 2016a, IATTC 2017), the establishment of EEZ-wide closures for sharks (Ward-Paige & Worm 2017), and the implementation of large no-take MPAs around the world (Lubchenco & Grorud-Colvert 2015, O'Leary et al. 2018). Despite these efforts, fish moving over large distances are sometimes thought to benefit little from closed areas as they only spend a limited amount of time within the protected areas and are exposed to fishing elsewhere (Hilborn et al. 2004, Grüss 2014). The argument that large pelagic predators roam too far to benefit from closed areas has been brought forward and discussed numerous times (Game et al. 2009, Davies et al. 2012). Indeed, a variety of movement patterns can strongly influence the effectiveness of spatial protection, including adult or juvenile migrations, long distance diffusive movements, the size of home ranges and density-dependent habitat choices usually triggered by prey availability (Grüss et al. 2011). Here we ask how suited large pelagic fish species are for spatial protection and how this relates to life history traits and management attributes, both of which can influence the suitability of area-based protection measures for these species.

Some informative data for the potential benefits of spatial protection derives from studies on other migrating pelagics, such as sea birds (Anderson et al. 2003, Trebilco et al. 2008, Péron et al. 2013, Young et al. 2015) and sea turtles (Maxwell et al. 2011, Scott et al. 2012). These studies demonstrate the importance of spatial protection during various vulnerable life stages such as nesting and breeding periods, as well as for juveniles. For seabirds, especially during egg incubation and chick rearing, foraging activities of adult birds such as albatrosses and petrels were found to concentrate in waters close to breeding colonies, amplifying interaction with fisheries operating in the area (Anderson et al. 2003, Trebilco et al. 2008). Sea turtles such as the Olive Ridley sea turtle (*Lepidochelys olivacea*) are known to remain in the nesting area for some time after laying eggs (Maxwell et al. 2011). Protected areas or spatial closures placed around nesting sites, breeding colonies and in relevant foraging habitats can directly contribute to decreased bycatch mortality of sea birds and sea turtles.

This review summarizes the current literature to determine which factors contribute to the efficacy and feasibility of spatial protection for large pelagic fishes. We consider both spatial fisheries closures implemented to aid the management of large pelagic fish stocks, as well as MPAs implemented to protect biodiversity at a broader scale. We employ the terms ‘closed areas’ or ‘spatial closure’ for both, acknowledging that fisheries closures and MPAs can be fundamentally different in purpose and design. Fisheries closures, both temporal and permanent, include area-based fisheries measures from gear restrictions to closures within a given extent, while MPAs are usually more restrictive regarding most or all types of industrial fishing, ranging from multi-use areas permitting low-impact, sustainable harvesting to no-take reserves (IUCN 2018a).

From this, we also discuss ways in which spatial fisheries management is currently used for these species and suggest how it may be further improved. We focus largely on information available for tuna (*Thunnus* and *Katsuwonus* genera), billfishes (swordfish, marlins [*Istiophoridae* family]) and pelagic sharks (such as great white shark [*Carcharodon carcharias*], blue shark [*Prionace glauca]*, or tiger shark [*Galeocerdo cuvier*]). The unifying characteristics of this diverse group of species is that they are highly mobile, they undertake long-distance horizontal movements through the pelagic environment, and they are currently exploited by fisheries.

Factors influencing suitability of spatial protection

Individuals of any given species are neither randomly, nor homogenously distributed through time and space. It follows that the vulnerability of large pelagics to fishing pressure varies with both location and life stage (Game et al. 2009, Juan-Jordá et al. 2013) and movement patterns as well as inter- and intraspecific behaviour strongly influence the response of pelagic fishes to particular management and conservation measures (Claudet et al. 2010). From a comparison of modelling, tagging, genetic, and fisheries-research related studies, a set of characteristics of the biology and life history of large pelagic fishes appears most influential regarding the efficacy of spatial protection; specifically, these include movement rates, as well as aggregation, philopatry, and restricted home ranges.

Movement rates

According to the majority of modelling studies, the main feature determining benefits of spatial protection for large pelagic fishes appears to be their movement rate within and between habitats—the higher their mobility, the lower the efficacy of spatial protection fixed in space (West et al. 2009, Grüss et al. 2011). This might be the case for bigeye tuna in the Central Pacific (Sibert et al. 2012) (discussed below), however, studies from the field are rare as dedicated spatial management exists for only 7 out of 40 stocks of major commercial tunas and billfishes examined by Pons et al. (2017). In addition to movement rates, the type of movement and the stage of life at which it occurs (Gaines et al. 2010) further influence the effects of spatial protection on migratory species. These include diffusive movement, dependence on home ranges, and density-dependent and independent movements, the latter including adult and ontogenetic migrations (Grüss et al. 2011).

The size of the area closed to achieve protection and lower fishing mortality is therefore related to dispersal rates and migration distances at different ages and can vary between 40-85% of the total area closed to obtain maximum yields for species with medium to high dispersal rates (Le Quesne & Codling 2009). Where detailed data are available for spatial planning, a trade-off of protected area size and area closed to fisheries can be achieved through networks of several smaller, well-placed and adequately spaced protected areas, specifically taking adult dispersal distances and larval connectivity into account (Palumbi 2004, Gaines et al. 2010). These areas are also known as "targeted MPAs" (Grüss 2014). However, in this context, enforcement plays a critical role, as multiple smaller reserves have a higher boundary-length to area protected ratio and infringements on the edges are likelier, potentially negating conservation benefits especially in areas with poor enforcement (Little et al. 2005). Thus, for remote and often data-poor pelagic areas, a closure of larger areas may be more efficient to increase fisheries benefits as well as stock rebuilding (Little et al. 2005, Stefansson & Rosenberg 2006).

Knowledge about a species' behaviour and movement as well as its predictability plays a major role in their protection (Table 6.1). For example, Atlantic, Pacific, and southern bluefin tuna return to well-defined spawning and feeding areas each year via known migratory routes (Block et al. 2001, ISC 2016b, CCSBT 2017). Likewise, for bigeye tuna in the Pacific Ocean, spawning and feeding movements within restricted home ranges are known (Kaplan et al. 2014). Adult bigeye tuna are caught primarily by longline vessels, while juveniles incur substantial bycatch by purse seiners fishing with fish aggregating devices (FADs). For predictable cases like these, closures such as the spawning ground closure in the Gulf of Mexico for Atlantic bluefin tuna are an option.

Other species like yellowfin tuna are opportunistic spawners requiring specific environmental conditions but are not necessarily bound to particular locations or routes (Reglero et al. 2014). Skipjack tuna (*Katsuwonus pelamis*) are not known to follow predictable spawning or feeding migration patterns, and environmental conditions are thought to play a primary role in dictating their movement patterns instead (Lehodey et al. 1997), complicating the design of targeted spatial closures based on their movement.

Modelling the outcomes of two kinds of spatial closures, the Chagos MPA as well as a large hypothetical fisheries closure in the Western Indian Ocean, demonstrates the importance of design and scale of spatial protection for "unpredictable" fish species (Dueri & Maury 2013). The Chagos MPA was found to have little effect on skipjack stocks due to strong seasonal variations of habitat conditions within the MPA. In contrast, a much larger hypothetical closure encompassed large parts of favourable habitat for skipjack tuna and was predicted to successfully stabilize spawning stock biomass (SSB). Remarkably, this closure also buffered fisheries yields lost due to the closure over a 20-year period, resulting in higher catches including the closure than without (Dueri & Maury 2013).

Despite the influence of mobility type and range, according to modelling studies highly mobile fish stocks within a system including a protected area still appear to be more resilient to collapse and fisheries yields higher over time than without (Apostolaki et al. 2002, Halpern et al. 2004, West et al. 2009, Dueri & Maury 2013, Buxton et al. 2014). Spatial protection is a vital component even in complex and highly developed fisheries management contexts, such as in Australia, where fish abundance and biomass continued to decline in sustainably managed fisheries without spatial protection but remained mostly stable with no-take marine reserves (Edgar et al. 2018). No-take marine reserves might further improve resilience against stock collapse for mobile species, especially under uncertain exploitation rates that may exceed target fishing mortality rates due to IUU fishing and bycatch, which emphasizes the role of MPAs as buffers against overexploitation under the precautionary principle (Lauck et al. 1998, Baskett & Barnett 2015). The timeframe over which these beneficial effects accrue varies with species biology and life history, as well as the influence of trophic interactions (Babcock et al. 2010, Moffitt et al. 2013). For target species, it is estimated that benefits of protection will result in higher target species biomass over one to two species generations (Sibert et al. 2012, Moffitt et al. 2013), with indirect effects on non-target species passing through trophic interactions and changes of ecosystems taking longer than that (Babcock et al. 2010).

Table 6.1 Characteristics of major commercial pelagic target species as defined in this study. IUCN red list status categories: NT = near threatened, EN = endangered, VU = vulnerable, LC = least concern, CR = critically endangered. Suitability for spatial management was assessed based on the specific life history attributes of each species.

Species	Defined migratory routes	Aggregation	Philopatry	RFMO harvest control rules	Targeted spatial management	Stock assessment	IUCN red list status*	Suitability for spatial management
Albacore tuna (*Thunnus alalunga*)	✓	✓	✓	North Atlantic stock	Mediterranean	✓	NT	High
Atlantic bluefin tuna (*Thunnus thynnus*)	✓	✓	✓	✗	Western Stock	✓	EN	High
Pacific bluefin tuna (*Thunnus orientalis)*	✓	✓	✓	✗	✗	✓	VU	High
Bigeye tuna (*Thunnus obesus*)	✗	✓	✗	Pacific stocks	Pacific stocks	✓	VU	Medium
Skipjack tuna (*Katsuwonus pelamis*)	✗	✓	✗	Pacific and Indian Ocean stocks	Pacific stocks	✓	LC	Medium
Yellowfin tuna (*Thunnus albacares*)	✗	✓	✗	Pacific stocks	Pacific stocks	✓	NT	Medium
Southern bluefin tuna (*Thunnus maccoyii*)	✓	✓	✓	✓	✗	✓	CR	High
Swordfish (*Xiphias gladius*)	some	✗	✓	✗	✓	✓	LC	Medium
Shortfin mako shark (*Isurus oxyrinchus*)	✗	✗	✓	✗	national shark sanctuaries	✓	VU	Medium

Species	Defined migratory routes	Aggregation	Philopatry	RFMO harvest control rules	Targeted spatial management	Stock assessment	IUCN red list status*	Suitability for spatial management
Blue shark (*Prionace glauca*)	?	✓	✓	✗	national shark sanctuaries	✓	NT	Medium
Silky shark (*Carcharhinus falciformis*)	✗	✗	?	✗	national shark sanctuaries	some	VU	Medium
Common thresher shark (*Alopias vulpinus*)	✗	✓	✓	✗	California	some (NOAA)	VU	Medium

*refers to global status, as listings may vary for regional stocks

Key sources: The Shark Trust 2018, Nakamura 1985, Compagno 2001, Hueter et al. 2004, Camhi et al. 2008, Froese & Pauly 2018, IUCN 2018

Aggregation, philopatry and restricted home range

Environmental heterogeneity as well as differences in phenotypes and behaviour ("behavioural polymorphism") —such as variations of movement rates of individual tuna (Sibert & Hampton 2003)— are assumed to govern the behaviour of individual fishes (Magurran 1993). While some fish disperse over wider areas or travel farther distances, others remain closer to areas where they have hatched, or return to aggregate in breeding, nursing, and feeding areas such as specific coastal regions or around hydrographic or bathymetric features such as seamounts (Holland & Grubbs 2007, Litvinov 2007).

Philopatry is the tendency to return to certain areas repeatedly and this behaviour has been demonstrated in many pelagic fishes. Chapman et al. (2015) reviewed more than 80 publications for residency and site fidelity in 31 shark species, including at least 6 migratory species. Based on tagging studies as well as DNA analyses, different philopatric behaviours, e.g. feeding site fidelity, are identified in large pelagic sharks such as tiger sharks and great white sharks (as reviewed by Chapman et al. 2015). Next to fidelity on local and regional scales, e.g. in South Africa and Australia (Pardini et al. 2001), white sharks can also exhibit strong repeated homing behaviours to specific places on fixed routes (Jorgensen et al. 2009). This includes regular visits to defined areas such as the so-called 'white shark café', an offshore region in the Northeast Pacific frequently visited by otherwise coastal great white sharks for presumably foraging and/or mating, and homing to very restricted coastal areas (Jorgensen et al. 2009). The same areas can be used by multiple species, as seen in Cleveland Bay, Australia, which is used as a communal nursery area by eight different shark species of the families Carcharhinidae (requiem sharks) and Sphyrnidae (hammerhead sharks) (Simpfendorfer & Milward 1993). Repeated returns to aggregate in a nursery area in the North Atlantic were also shown for oceanic blue sharks with high abundances of juveniles and several tagged adults frequently returning to the same area (Vandeperre et al. 2014).

Site fidelity, aggregation, and restricted movement patterns were also found for several tuna species such as Atlantic bluefin tuna, which has an eastern stock that spawns in the Mediterranean Sea and a western stock that returns annually to spawn in the Gulf of Mexico—yet mixing occurs in both these sites as well as at foraging grounds (Block et al. 2005). Similarly, although they make trans-oceanic migrations between Japan and Mexico, Pacific bluefin tuna also have specific spawning grounds in the East China Sea and the Sea of Japan (Schaefer 2001). Southern bluefin tuna spawn off Java in the Indian Ocean but juveniles (2-5 years old) undertake seasonal migrations to spend the austral summer in the Great Australian Bight and the winter off New Zealand or South Africa and older fish disperse widely across the western Atlantic and the Indian Ocean to the Tasman Sea (Hobday et al. 2015).

While some individuals leave and occasionally come back, others remain in the same region throughout their lives: Some populations (or parts thereof) of yellowfin, skipjack, and bigeye tuna exhibit restricted movement ranges, low dispersion levels and/or high site fidelity (Schaefer et al. 2011, 2014, Wells et al. 2012). Around the Hawaiian Islands, yellowfin tuna, especially juveniles, were found to have displacement distances of only 50 km (Itano & Holland 2000, Adam et al. 2003) and high retention rates with 91% of sub-adult yellowfin tuna sampled in the Hawaiian Islands originating from a known spawning ground in the area (Wells et al. 2012). These and other tagging studies provide important baseline data on movement patterns, which can then inform on vulnerability to fishing as well as the potential for protection of large pelagic predators. Results from tagging can be especially useful with regard to the identification of possible site fidelity of subsets of populations that might benefit more from spatial protection than individuals exhibiting higher mobility. These 'lazy' semi-resident fish with low movement rates are assumed to be favoured by spatial protection, establishing populations inside the protected area. They possibly contribute to population-level behavioural changes such as decreased movement rates of fish over time, especially under high fishing pressure (Miethe et al. 2008, Mee et al. 2017). Though unstudied in the field so far, increased residency might positively affect stock sizes and size at maturation (Miethe et al. 2008) within a protected area but potentially negatively affect 'spillover' of fish catches into adjacent areas, stock connectivity, as well as genetic resilience to environmental changes (Dawson et al. 2006).

As more data on distributions, home ranges as well as area and habitat use of large pelagic fish are available from a variety of field studies it becomes clear that many migratory species move along predictable, often shared migration routes such as transition zone chlorophyll fronts (Polovina et al. 2001, Block et al. 2011), aggregate in specific places or during certain life stages, and exhibit homing behaviour or site fidelity (Table 6.1). The higher the predictability of these behaviours, the more targeted conservation approaches such as spatial protection can be. The reduction of area-specific threats especially in frequently used habitats might therefore lead to disproportional benefits compared to the size of the area protected (Game et al. 2009). This is especially important as predictable occurrences, especially aggregations, are often preferentially targeted by fisheries, rendering these species more vulnerable to overfishing (Litvinov 2007).

Some targeted spatial protection measures to protect vulnerable aggregations and life stages are already in place on national and international levels. A few known aggregation and spawning sites are included in MPAs (e.g. the Phoenix Islands Protected Area, Kiribati), whereas other sites are subject to specific spatial management and regulations by countries and RFMOs, such as the bluefin tuna spawning grounds in the Gulf of Mexico and the Mediterranean Sea (Fig. 6.1 and Table 6.1) and fishing gear restrictions to protect juvenile fish e.g. bigeye tuna (see below). Oceanographic features such as seamounts and ridges are an increasing focus of spatial protection (Clark et al. 2011) and several MPAs such as the Charlie-Gibbs MPA and the Josephine Seamount MPA in the North Atlantic, and the SGaan Kinghlas-Bowie Seamount MPA off the Canadian West coast have been specifically designed to protect them. However, while known migration routes, as well as spawning and aggregation areas appear low-hanging fruit for conservation measures, the management and protection of large pelagic fish is often not the primary objective. For example, the afore mentioned Cleveland Bay, Australia, a known shared nursery for several shark species, is a designated dugong protection area but lacks shark-specific protection and fishing is permitted with a few exceptions such as net fishing. Other known migratory routes, such as the Cocos-Galápagos Migratory Pathway between the Galápagos Islands the Cocos Island Marine Reserves used by a variety of sharks, rays, and turtles, remain unprotected, too, despite knowledge of intense illegal fisheries in the area (Kyne et al. 2012, White et al. 2015). While some High Seas areas are subject to regulations by several RFMOs (see below and Fig. 6.1)establishment and enforcement of spatial closures remains a challenge, a fact that is addressed in the current negotiations on a High Seas treaty by the United Nations.

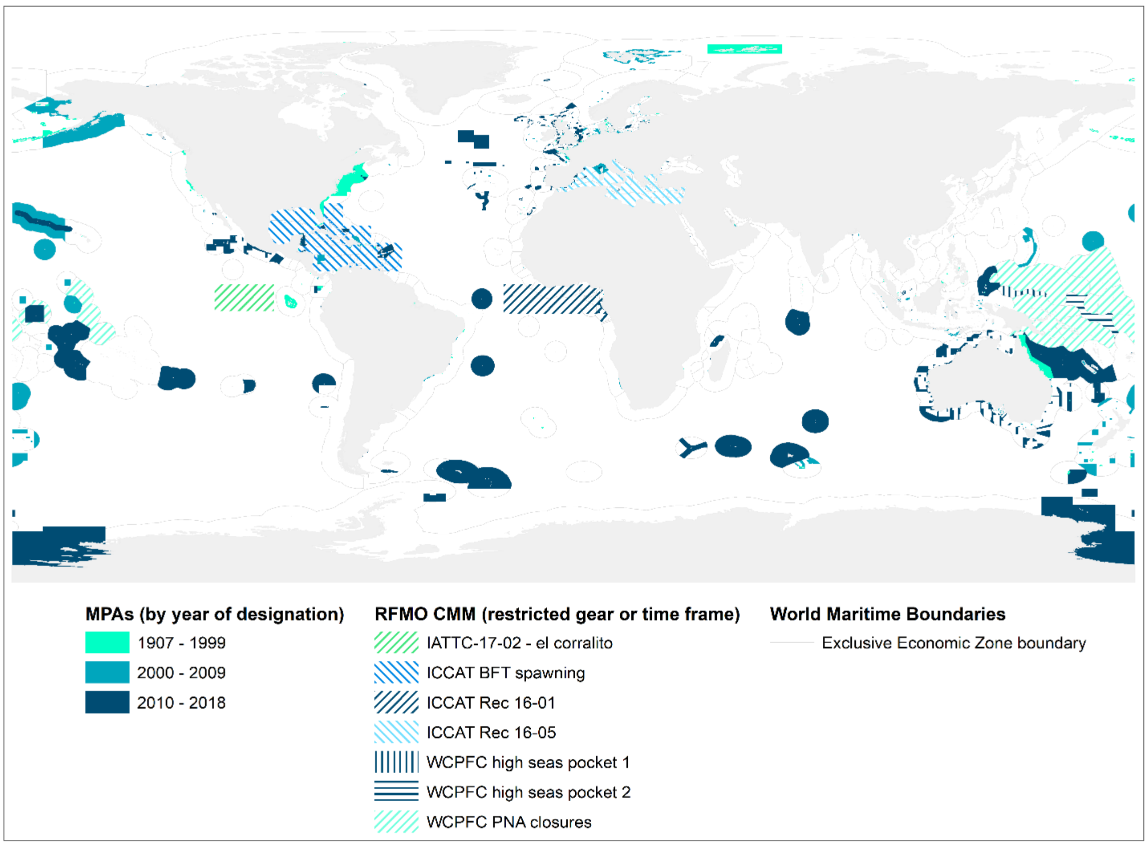

Fig. 6.1 Locations of major spatial closures around the world. Marine protected areas (MPAs) are colour coded by year of establishment as taken from the World Database on Protected Areas (via protectedplanet.net). Active spatial Conservation and Management Measures (CMMs) of Regional Fisheries Management Organizations (RFMOs) are shown in hatched patterns. Note that these are typically seasonal or gear specific closures. The temporary closure of the IATTC Convention Area in the Eastern Tropical Pacific (IATTC 17-02) is not included; it was classified as a fisheries management measure as it comprises the whole convention area. Marine Protected areas shown here cover about 7%, and RFMO CMMs cover about 7.4% of total ocean area.

Fisheries benefits of spatial protection for large pelagics

When protected areas or closures are planned, the benefits of conservation and protection of marine life are often measured against costs and losses to stakeholders such as fisheries. In this context it is important to consider that fisheries can potentially benefit both directly and indirectly from spatial closures. A well-documented benefit is the spillover of adult fish from the protected area, contributing to higher, more stable catches in surrounding areas (Russ & Alcala 2011, Kerwath et al. 2013, Boerder et al. 2017). Spillover of larvae ("recruitment effect") can additionally replenish populations in adjacent waters after a time lag (Christie et al. 2010). Spatial closures can also be used to reduce bycatch, which is especially important for fisheries that are subject to bycatch limits and caps (Diamond et al. 2010, O'Keefe et al. 2014, Little et al. 2015).

Two main factors influence the magnitude of benefits of spatial closures for fisheries: the state of associated fisheries, namely current fishing mortality (F) relative to the maximum sustainable rate of fishing mortality (F_{MSY}), and the dynamics of fishing fleets. Positive effects of protected areas on fish stocks and catches appear less strong when fisheries in the wider area are well-managed and catches are close to MSY (Guénette & Pitcher 1999, Apostolaki et al. 2002, Hilborn et al. 2006, Buxton et al. 2014). In these cases, the value of lost catch due to area closure is likely not outweighed by benefits of the protected area to fisheries such as spillover, especially if effort is kept constant. At the same time, however, catch losses due to area closures are less severe for large pelagics as their range usually greatly exceeds the closed areas and they can be caught somewhere else (Apostolaki et al. 2002). However, if fishing mortality (F) outside the reserve is too high (above F_{MSY}) and the gradient of fish abundance from inside to the outside of a reserve is large, fish stocks and ill-managed fisheries might benefit significantly from larval and adult spillover from the protected area providing recruitment subsidies and replenishing populations outside (Gerber et al. 2003, Le Quesne & Codling 2009, Green et al. 2015).

Adaptation of fishing effort or capacity to the area open to fishing is deemed essential to avoid a 'squeeze factor', concentrating the same amount of fishing in a smaller area (Halpern et al. 2004) and potentially affecting other species and habitats (Greenstreet et al. 2009). Even wide-ranging large pelagics may be negatively impacted by displaced fishing effort (Baum et al. 2003). In response to the closures of the High Seas pockets between the Pacific Island EEZs in the Western Central Pacific, purse seining effort for skipjack and bigeye tuna simply redistributed to EEZ surrounding the closures and no beneficial effect of the closures for bigeye tuna was detectable (Sibert et al. 2012). Around Cocos Island National Park, several target species such as the scalloped hammerhead shark (*Sphyrna lewini*) are declining despite protection, largely due to a mix of changed community compositions as well as unabated heavy fishing both inside and outside the protected areas (Kyne et al. 2012, White et al. 2015). These consequences can also be unintended: Simulations of closures of large areas with different protection goals (e.g. protection of sea turtles or coastal shark species) predicted a negative impact of displaced fishing effort on a variety of pelagic shark and finfish species (Baum et al. 2003).

However, an adequate combination of spatial protection and fisheries management can have positive effects as found by Ward-Paige et al. (2010) in the Indian, Atlantic, and Pacific Ocean, where several large pelagic sharks such as tiger, silky (*Carcharhinus falciformis*) and bull sharks (*Carcharhinus leucas*) were more frequently sighted in areas with low human population or in regions with well-regulated fisheries or enforced marine reserves.

While the importance of combining spatial protection measures with other fisheries management tools such as effort controls seems clear, adaptation of fishing effort (input control) is not always achievable. In these cases, output controls such as TACs might be an alternative. Pons et al. (2017) found that enforced TACs in combination with minimum size regulations and spatial closures yielded benefits to rebuild major commercially exploited tuna and billfish stocks. For bigeye tuna, for example, a combination of closed areas for certain gear types (longlines) as well as fisheries management (prohibition of the use of FADs) was effective to increase adult biomass due to a simultaneous reduction of fishing mortality of both sexually mature adults in longline fisheries and juveniles in purse seine fisheries on FADs (Sibert et al. 2012). Scenarios focussing on the use of total fisheries closures showed strongest effects, however, when fisheries management controlled for (eliminated) displaced fishing effort, amounting to 10-25% increase of adult biomass over the entire range of Pacific bigeye tuna.

For some species, conventional fisheries management tools such as catch and gear restrictions might suffice for adequate management (Hilborn et al. 2006). Such cases potentially apply for highly target-specific fisheries with little bycatch (Hilborn et al. 2004), and for species for which information on predictable aggregations or migratory patterns is lacking, such as silky sharks (Table 6.1). However, the inclusion of protected areas in fishery management plans is desirable as protected areas can buffer against management uncertainties (Stefansson & Rosenberg 2005, 2006), such as those caused by limited scientific knowledge and environmental ambiguity (Lauck et al. 1998). Likewise, the weaker and more uncertain overall fisheries management is, the more important spatial protection becomes for protection of habitats, sensitive species, and their genetic variability to increase resilience (Roberts et al. 2005). While marine reserves cannot safeguard against all stressors imposed on a given stock, the reduction of at least a few threats such as extractive and non-extractive uses and habitat destruction lessens cumulative pressure (Game et al. 2009). Especially in areas where conventional fisheries management like catch controls is not applicable spatial protection might be the most, or even only, effective means to achieve benefits (Hilborn et al. 2004).

How are closures currently used for pelagic fish?

A number of MPAs and spatial fisheries closures have been established in all ocean basins. MPAs currently cover about 7% of global ocean surface area and range from small, coastal MPAs to large, offshore MPAs encompassing up to 2 million square kilometres. Fisheries management closures, both seasonal and permanent time/area or gear specific closures, are common on national and multilateral levels within EEZs and RFMO convention areas and currently cover about 7.4% (Fig. 6.1). While some protected areas and closures date back five decades and more, most have been created and established since the early 2000s. However, their effects on large pelagic fish remain often not well documented. Here we summarize what is known from a number of case studies (Table 6.2).

Some differences are notable in the use of MPAs versus spatial closures. While the establishment of MPAs is typically fixed in space and time and guided by static habitat features such as reefs or seamounts, pelagic closures can be more dynamic and may be changed and adapted to specific conservation goals over time (Hyrenbach et al. 2000, Game et al. 2009, Grüss et al. 2011). Given reliable data on spatial and temporal distribution and area use of each species these closures could be fully dynamic in space and time (Hyrenbach et al. 2000, Block et al. 2005, Game et al. 2009, Grüss et al. 2011, Maxwell et al. 2015, Dunn et al. 2016). Another difference is that MPAs are almost always unilateral, i.e. declared by individual countries within their EEZs, whereas spatial closures are often (but not always) multi-lateral and managed primarily at the RFMO level (Fig. 6.1). Measures to reduce fishing mortality for particular species, especially for their juveniles, and in key spawning and aggregation areas appear to be the primary means of protecting pelagic fish at present.

Table 6.2 Selected empirical examples documenting benefits of different types of spatial closures on large pelagic fish. Details are explained in the text.

Type	Area	Time frame	Documented benefits	Sources
Large-scale MPA	Galápagos Marine Reserve	since 1998	elevated yellowfin tuna catch rates close to reserve, increased yellowfin and skipjack productivity in and around reserve	Boerder et al. 2017, Bucaram et al. 2018
MPAs	Various (87 sites worldwide)	since 2006	increased biomass and diversity of large pelagic fishes	Edgar et al. 2014
MPA Networks with strong fisheries management	Florida, Bahamas, U.S. Virgin Islands	1993-2008	only Caribbean countries with large sharks abundant	Ward-Paige et al. 2010, Graham et al. 2016
EEZ-wide shark sanctuary	Shark Sanctuaries (15 countries)	since 2009	reduced shark fishing and slower shark population declines	Ward-Paige & Worm 2017
Unilateral fisheries closure	Baja California billfish closure	1977-1980, 1984-1985	22% increase of striped marlin abundance	Squire & Au 1990, Jensen et al. 2010
Unilateral fisheries closure	US Atlantic swordfish closures	since 1999	reduction of swordfish bycatch contributed to stock rebuilding	NMFS 2006
IATTC closure	Purse seining and *'el corralito'* closures	since 2004	reduction of fishing mortality (primarily bigeye tuna)	Xu et al. 2018
WCPFC restriction	FAD closure	since 2008	reduction of juvenile bigeye tuna bycatch	Hanich et al. 2010, SPC-OFP 2012
ICCAT closure	Mediterranean swordfish closure	since 2011	reduction of total catch of adult and juvenile swordfish	ICCAT 2016a

Unilateral measures

Unilateral measures for protection of pelagic fish can include MPAs or time/area closures depending on specific context and documentation of their effects is available in a few cases. The Galápagos Marine Reserve (GMR) is a 133,000 km^2 marine reserve established in 1998 by the Ecuadorian government with the aim of preventing industrial fishing to protect the rich marine biodiversity and endemism of this Pacific island chain. The marine reserve also protects a presumed tuna nursery (Kliffen & Berkes 2015). Positive effects of this closure are seen through fleet behaviour aggregating in the direct vicinity of the reserve ('fishing the line') and achieving higher catch rates compared with surrounding areas (Boerder et al. 2017) (Chapter 3). Commercial tuna fishermen are aware of positive reserve effects for tuna stocks and preferably fish close to the reserve boundaries to maximize benefits (Kliffen & Berkes 2015). As documented from a combination of on-board observer data as well as Automatic Identification System (AIS) vessel tracking data, four times more purse seine sets for tuna were deployed within 20 km from the reserve boundaries compared to the rest of the study area (400 km), presumably to benefit from spillover (Boerder et al., 2017). In addition, after establishment of the GMR, catch, effort and catch-per-unit-effort (CPUE) patterns in the wider area have shifted closer to the reserve boundaries where overall declining catch trends of the three major tuna species appear to be buffered by reserve benefits. These effects were most pronounced for yellowfin and skipjack tuna, which show increased productivity both inside and around the GMR (Bucaram et al. 2018).

No-take MPAs that are large (>100km^2), older (>10 years), well-enforced, or in isolated locations also have been shown to have predictable conservation benefits for large fish, which increase both in abundance and diversity, according to a comprehensive meta-analysis of 87 MPAs worldwide (Edgar et al. 2014). Effects were especially pronounced for sharks which doubled in abundance across all MPAs and increased up to 20-fold in areas that had all of the above-mentioned features. The strongest effects for sharks were observed as a function of area isolation, size and age, in that order (Edgar et al. 2014). A caveat for this study is that many of the species were reef-associated, but a number of pelagic sharks and jacks were also present.

Protected area networks in connection with improved fisheries management were also thought to be important in allowing large sharks to persist in some abundance in Florida, the Bahamas and the US Virgin Islands, in notable contrast to the remainder of the Caribbean (Ward Paige et al. 2010). Electronic tagging studies later confirmed that established MPAs in Florida and the Bahamas are indeed being used by, and provide protection to, great hammerhead (*Sphyrna mokarran*) and tiger sharks. However, while these species benefitted from spatial protection, bull sharks (*Carcharhinus leucas*) did not (Graham et al. 2016).

Fisheries closures can provide similar benefits for large pelagic fish. Mexico established a series of such closures for longline fisheries in the Mexican Pacific EEZ in Baja California between 1977-1980 and 1984-1985 to reduce commercial fishing mortality of billfishes. Jensen et al. (2010) constructed a stock reduction analysis model based on Japanese longline fishery data for striped marlin and were able to confirm earlier observations made by Squire and Au (1990) from raw CPUE data documenting increases of abundance of striped marlin up to 22% in relation to the closures. Striped marlin in this area likely benefitted from the closures due to restricted movement patterns of a large proportion of the local stock but data on current stock status is lacking (IATTC 2015).

Likewise, the United States National Marine Fisheries Service has implemented a series of unilateral time and area closures on the Atlantic coast since 1999 in its domestic tuna, shark and swordfish fisheries. These closures have successfully reduced bycatch in these fisheries and contributed to the recovery of the Atlantic swordfish stock although other billfishes such as the Atlantic white marlin continue to be overfished (NMFS 2006). On the Pacific coast, the entire U.S. EEZ is closed to industrial pelagic longlining for tunas and swordfish— a measure that is also meant to reduce bycatch of common thresher shark (*Alopias vulpinus*), sea turtles, and marine mammals. Drift gillnetting for swordfish and sharks is prohibited in certain parts of the U.S. EEZ in order to reduce bycatch of these and other coastal species (Pacific Fishery Management Council 2018).

As of 2018, several countries such as Palau, the Marshall Islands and the Maldives, have established 17 shark sanctuaries in parts or the whole of their EEZ, covering nearly 20 million square kilometres. Commercial and sometimes also small-scale fishing for sharks is typically banned in these areas, combined with a ban of retention, possession, and trade of bycaught sharks (Ward-Paige & Worm 2017). While the creation of specific sanctuaries for sharks has received international attention, their effectiveness remains uncertain due to difficulties in monitoring and enforcement (Davidson 2012) as well as bycatch mitigation (Ward-Paige 2017).

Not all closures are static in space and time, increasingly the feasibility of dynamic spatial management measures is discussed (Maxwell et al. 2015, Dunn et al. 2016) and attempted, mostly in the context of avoiding bycatch of threatened species such as North Atlantic right whales (*Eubalaena glacialis*) (Van Parijs et al. 2009) or loggerhead sea turtles (*Caretta caretta*) (Howell et al. 2008). Here, dynamic spatial closures have proven successful at reducing entanglement of these species in certain fishing gears. Siting of these dynamic closures is guided by visual, acoustic and thermal habitat observations. For tunas, dynamic closures have been applied in a similar fashion to avoid bycatch of Southern bluefin tuna (Hobday & Hartmann 2006, Hobday et al. 2010).

RFMO measures

In addition to unilateral spatial management and MPA establishment, four of the five tuna RFMOs have also included spatial closures as a tool for managing heavily fished target stocks. The earliest record of this includes temporal closures to purse seining for yellowfin tuna in the Inter-American Tropical Tuna Commissions' (IATTC) Yellowfin Regulatory Area from 1966-1978 and in 1999-2001 (Table D.1). These closures were viewed as beneficial for restricting fishing effort on yellowfin tuna, although their applicability and the subsequent uptake of similar measures in the early 2000s for bigeye tuna were deemed less successful given initial challenges with determining an appropriate TAC for this species and the relatively low catch of bigeye tuna by purse seiners at the time (IATTC 2006).

Since 1993, the International Commission for the Conservation of Atlantic Tunas (ICCAT) has prohibited directed fishing for Atlantic bluefin tuna in their Gulf of Mexico spawning grounds (ICCAT 1993), although independent research suggests that incidental catch of this species was occurring through the mid-2000s (Block et al. 2005). Although this CMM is the oldest RFMO spatial measure adopted still in place today, its success remains unknown as there is currently a high degree of uncertainty around the state of the stock as a whole (ICCAT 2017a). Furthermore, given additional uncertainty around the exact spawning location of the eastern Atlantic bluefin tuna stock, no additional spatial protection for this species exists in the Mediterranean Sea. Although spawning sites of both other bluefins are known, no CMMS have been adopted in relation to spatial management for these species by members of the Western and Central Pacific Fisheries Commission (WCPFC) or of the Commission for the Conservation of Southern Bluefin Tuna (CCSBT).

ICCAT, WCPFC and IATTC all have spatial measures in place as part of larger fisheries management plans for the key tuna stocks under their jurisdictions: skipjack, yellowfin, and bigeye tuna. Given the high degree of juvenile tuna bycatch incurred through the use of FADs, these devices have been prohibited in specifically defined areas during certain months of the year by both ICCAT and the WCPFC (Fig. 6.1). WCPFC members have also adopted increasingly stricter spatial management measures over the last decade, largely in conjunction with the fishing regulations laid out by Parties to the Nauru countries with regard to access to their EEZs (Hanich et al. 2010). Due to concerns over elevated fishing mortality of juvenile bigeye tuna, a variety of closures to purse seining with FADs were adopted as part of CMM 2008-01 with explicit requirements that fishing states refrain from transferring effort from these closures to other fishing areas. This measure was largely successful in substantially reducing bycatch of juvenile bigeye tuna (SPC-OFP 2012). Despite this, economic losses were minimal as the reduction in volume was offset by the higher value of larger individuals landed as more fishing occurred by unassociated sets, which catch larger fish (SPC-OFP 2012). Since their original implementation, there has been a temporal extension of these FAD closures and, presently, overfishing of the bigeye tuna stock in the Western Central Pacific is not occurring (McKechnie et al. 2017).

The IATTC has arguably the most extensive spatial management measures: a three-month closure to all industrial purse seining within the Convention Area, as well as a one-month spatial closure in a region known as *el corralito* during the fall (Fig. 6.1). Variations of both measures were first adopted by IATTC members in 2004 and have since been expanded both spatially and temporally (i.e., 59 days in 2010 to 72 days at present). These closures are viewed as a means of reducing fishing mortality (primarily of bigeye tuna) by controlling capacity and, in combination with other management measures, these closures are believed to have met these objectives between 2005-2009 (Xu et al. 2018). However, overcapacity of the Eastern Tropical Pacific purse seine fleet remains a challenge and the bigeye tuna stock is currently subject to overfishing (Xu et al. 2018).

The IOTC has implemented two spatial management measures for target tunas, although neither of these are still active. While these measures were adopted to decrease effort on bigeye and yellowfin tuna, they did not appear sufficient for achieving these aims, likely as a result of uncertainty around stock dynamics as well as a redistribution of fishing effort outside of closed areas (IOTC Scientific Committee 2010).

Swordfish is the only billfish species for which spatial management measures have been adopted at the RFMO level. Directed fishing and retention of this species is prohibited in the Mediterranean Sea for three months annually. As a result of the establishment of the first version of this CMM in 2011, there was a significant reduction in total swordfish catch as well as a 50% decrease in the volume of juveniles caught relative to the 2000s. As the majority of juvenile swordfish bycatch occurs during the fall, an additional two-month closure to the Mediterranean albacore tuna longline fleet was established in 2016 and the effectiveness of this new measure will be evaluated in the near future (ICCAT 2016a).

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Options for improving spatial protection of pelagic fish

The continued uptake—and proliferation—of spatial closures in CMMs at the RFMO level suggests spatial management is seen as a valuable complement to other input controls used to control fishing capacity and catch of tunas and other large pelagic fishes. Still, all CMMs that include a spatial element were developed with target species (i.e., tunas and swordfish) as the focus, which suggests there is room for improvement when it comes to adopting spatial measures to mitigate bycatch or ensure the sustainable extraction of the other large pelagics under their jurisdiction. Many CMMs for bycatch species have been developed and amended in an *ad hoc* fashion, so first determining areas of special concern for both target tunas and associated large pelagics would be beneficial in developing subsequent spatial management plans for these species (Juan-Jordá et al. 2013). For target species, of the four species identified in Table 6.1 as having a 'high' suitability for spatial management due to their species attributes, specific area-based fishing measures exist for only two: albacore (Mediterranean stock) and Atlantic bluefin tuna (western stock). Yet, the effectiveness of the closure for western bluefin tuna spawning in the Gulf of Mexico is debatable since bycatch of these species in other fisheries continues (Table D.1) and the closure to albacore tuna fishing was devised as a means of addressing swordfish bycatch, not albacore tuna mortality. Bearing these circumstances in mind, both southern and Pacific bluefin tuna, as well as Atlantic bluefin tuna—all of which are considered depleted—may benefit from stronger targeted spatial management measures. Since all of these species exhibit philopatry, it seems that improved protection of spawning sites, even temporary, could be highly beneficial. However, as these locations occur within the national waters of specific countries (e.g. Japan), they would only succeed with the support and oversight of these states. Also, as is evidenced by Atlantic bluefin tuna, ensuring bycatch of these species within closed areas is avoided is also essential.

Additionally, given shifts of species distributions and migratory routes with changing environmental conditions (Hazen et al. 2013, Morley et al. 2018) the incorporation of dynamic components into adaptive marine spatial planning and fisheries management is becoming increasingly important (Maxwell et al. 2015, Monllor-Hurtado et al. 2017). While dynamic management requires substantial prior knowledge and complex habitat prediction models as well as precise enforcement, the higher precision of area closures tailored to species' presence and absence reduces the amounts of area and length of time of closures (Maxwell et al. 2015, Dunn et al. 2016). For these cases, novel satellite-based tools such as multi-sensor remote sensing (e.g. synthetic-aperture radar [SAR]) as well as vessel tracking such as national vessel tracking system [VMS] and AIS) from space can support oceanographic data collection and monitoring as well as improve enforcement even on the High Seas (Turner et al. 2003, Zainuddin et al. 2006, Dunn et al. 2018, Kroodsma et al. 2018).

While comprehensive monitoring tools to track fishing fleets are now available (Kroodsma et al. 2018) and should aid in improving the visibility of fishing activity around the world, fundamental challenges in policy design and implementation—and the associated enforcement capabilities along those policies—remain. In 2015, the UN General Assembly called for the development of an international legally-binding instrument under UNCLOS to address the conservation and sustainable use of marine biological diversity of areas beyond national jurisdiction (Resolution 69/292). However, as of the most recent preparatory committee meeting (July 2017) the degree to which fisheries management is explicitly included remained uncertain although area-based management strategies (including MPAs) have been identified as a key topic for discussion (current documentation can be found under http://www.un.org/depts/los/biodiversity/prepcom.htm). The evidence presented in this review support this and calls for a coordinated approach in which spatial closures are adopted in conjunction with relevant fisheries management regulations such as effort controls, particularly to avoid effort displacement to unprotected areas.

Conclusion

While close to 15% of global ocean area is now under some form of targeted spatial management (Fig. 6.1 and Table D.1) and many pelagic fish appear suitable for spatial production based on their life history (Table 6.1), the various effects of these measures on large pelagic fishes are documented in comparatively few case studies (Table 6.2) and may be difficult to generalize. Yet, their propensity to aggregate, as well as the defined migratory patterns and philopatry observed in certain large pelagic species suggest that highly migratory species can benefit from targeted, well-designed spatial protection, especially in spawning or nursery areas or of geomorphological features that aggregate species such as seamounts and thermal fronts, and for critical life stages such as juvenile fish. Additionally, spatial protection can be more beneficial when stocks are overfished or subject to high bycatch rates. In conjunction with effective, transboundary fisheries management regimes, spatial protection measures can provide additional benefits in terms of increased habitat quality, increased resilience to stock collapse, insurance against management errors, and protection of non-target species and associated biodiversity. Next to unilateral spatial protection measures, RFMO member states have implemented spatial management for several highly migratory target species, although the degree to which vulnerable life stages and areas (e.g. spawning sites) are protected appears inadequate and significant potential regarding spatial protection measures for these migratory fishes still exists. The results summarized in this review may be beneficial for policy discussions such as the recent United Nations treaty targeting areas beyond national jurisdiction, which seeks to better regulate activities and protect biodiversity on the High Seas.

Appendix D

Table D.1 Spatial management measures adopted by the tuna RFMOs. Note: CMMs with an asterisk (*) are no longer active. LL stands for longline, PS for purse seine, FAD for fish aggregating device, HS for High Seas.

RFMO	Species	Measure	Summary	Results	Source(s)
ICCAT	Atlantic bluefin tuna	Rec. 93-05* (replaced by 98-07, 06-06, 08-04, 10-03, 12-02, 13-09, 14-05,16-08 and 17-06)	Spatial element as part of a series of adopted fishing measures: • No directed fishing for Atlantic bluefin in spawning grounds (i.e. Gulf of Mexico)	• Effectiveness of spatial component not explicitly discussed in 2017 ICCAT Scientific Committee report • Independent research suggests bluefin were still caught incidentally in Gulf LL fisheries up to 2003	ICCAT (1993), ICCAT (2017c), Block et al. (2005)
ICCAT	Bigeye tuna	Rec. 04-01* (replaced by 08-01, 11-01)	Spatial closure as part of a series of measures to address juvenile bigeye catch: • From 2005-2008, no fishing by PS (all types) or bait boats, 20º – 10ºW, 5ºN-0ºS (1 Nov – 30 Nov)	• Not effective at reducing the mortality of juvenile bigeye, and minimal reduction in yellowfin mortality due to the redistribution of fishing effort into adjacent areas	ICCAT (2004), ICCAT (2017c)
ICCAT	Swordfish	Rec. 07-01* (replaced by Rec 09-04, Rec. 11-03)	To reduce fishing mortality and increase stock biomass: • For 2008, prohibition on all fishing of swordfish, Mediterranean Sea, 15 Oct – 15 Nov	• Closure appeared to be beneficial and was thought to be able to move the stock toward a biomass that could support MSY (only a closure longer than 4-months predicted to rebuild to 1980s SSB levels)	ICCAT (2007), ICCAT (2010)

RFMO	Species	Measure	Summary	Results	Source(s)
ICCAT	Bigeye, yellowfin and skipjack tuna	Rec. 11-01* (replaced by 14-01, 15-01, 16-01)	Spatial closure (FAD specific) adopted to mitigate catch of juvenile tunas: • For 2013 onward, no PS fishing with FADs, African coastline to 10ºS, 5ºW- 5ºE (1 January- 28 February)	• Included in evaluation as per Rec. 04-01	ICCAT (2011a), ICCAT (2017c)
ICCAT	Swordfish	Rec. 11-03* (replaced by 13-04, 16-05)	To further address overfishing of swordfish and the bycatch of juvenile swordfish: • From 2012 onward, prohibition of catch or retention of swordfish (all gears), Mediterranean Sea (1 Oct – 30 Nov as well as one month between 15 Feb – 31 Mar)	• Significant reduction in total swordfish catch (among the lowest in three decades), catch of juvenile swordfish down 50% relative to 2000s • High bycatch of swordfish in autumn albacore LL fishery remains	ICCAT (2011b), ICCAT (2016b), ICCAT (2017c)
ICCAT	Bigeye, yellowfin and skipjack tuna	Rec. 16-01	Replaces Rec. 11-01 and all subsequently adopted recommendations (see previous) and modifies boundaries of spatial closure, i.e.: • No PS fishing with FADs, African coastline to 20ºW, 5ºN- 4ºS (1 January- 28 February)	• Effectiveness to be evaluated in 2018 at the latest and possible alternative areas/times also to be considered	ICCAT (2016c)

RFMO	Species	Measure	Summary	Results	Source(s)
ICCAT	Swordfish	Rec. 16-05 (replaces 13-04)	As part of a suite of measures to assist in rebuilding the Mediterranean swordfish stock to 2031: • Prohibition of swordfish catch or retention, Mediterranean Sea (1 Oct – 30 Nov as well as one month between 15 Feb – 31 Mar *or* 1 Jan – 31 Mar inclusive) • Closure to albacore LL fleet, Mediterranean Sea (1 Oct – 30 Nov)	• Effectiveness to be evaluated by Scientific Committee	ICCAT (2016a)
ICCAT	Atlantic bluefin tuna	Rec. 17-06	Retains original spatial component (as above) as part of comprehensive rebuilding plan for western Atlantic bluefin	• No additional analysis subsequent to Rec 93-05 but evaluation of efficacy of current measure to be provided by Scientific Committee (based on results, resolution may be amended in future)	ICCAT (2017a)

RFMO	Species	Measure	Summary	Results	Source(s)
IATTC	Bigeye, yellowfin and skipjack tuna	C-04-09* (replaced by C-06-02, C-09-01, C-11-01, C-13-01, C-17-01/02)	Multiple measures implemented to control fishing capacity, spatial components: • Annual PS closure, coastline to 150°W, 40°N-40°S (fleets may choose one of two time periods: 1 Aug – 11 Sept or 20 Nov – 31 Dec)	• Closures viewed as an effective means of offsetting total PS capacity in order to meet management objectives i.e. as of 2015: reduce fishing mortality to a level corresponding with MSY for all tuna stocks) • "Recovering trend for bigeye in the EPO during 2005-2009, subsequent to IATTC tuna conservation resolutions initiated in 2004" (although overfishing currently occurring)	IATTC (2004), IATTC Scientific Committee (2015), Xu et al. (2018)
IATTC	Bigeye, yellowfin and skipjack tuna	C-17-02	Replaces C-04-09 and all subsequently adopted CMMs (see previous) and extends PS closures until 2020, i.e.: • 72-day closure to all industrial PS within IATTC Convention area (fleets may choose one of two time periods: 29 July - 8 Oct or 9 Nov - 19 Jan) • Annual spatial closure for PS, 96º-110ºW and 4°N-3°S (area known as *el corralito*) (9 Oct – 8 Nov)	• No additional analysis subsequent to C-04-09	IATTC (2017)

RFMO	Species	Measure	Summary	Results	Source(s)
IOTC	Bigeye, yellowfin and skipjack tuna	Res. 10/01* (replaced by Res. 14/02)	2-year measure (2011-2012) aimed at decreasing fishing pressure on target stocks, in particular bigeye and yellowfin (specific targets undefined). Spatial components: • LL closure, 0-10ºN and 40-60ºE (1 Feb – 1 Mar) • PS closure, 0-10ºN and 40-60ºE (1 Nov – 1 Dec)	• Difficult to ascertain effectiveness due to uncertainties around movement rates of fish and potential redistribution of fishing effort	IOTC (2010), IOTC Scientific Committee (2010)
IOTC	Bigeye, yellowfin and skipjack tuna	Res. 12/13*	Extension of spatial components defined in Res. 10/01 (until 2014), no subsequent CMMs include spatial management component for IOTC stocks	• No additional analysis subsequent to Res. 10/01	IOTC (2012)

RFMO	Species	Measure	Summary	Results	Source(s)
WCPFC	Bigeye, yellowfin and skipjack tuna	CMM 2008-01* (replaced by CMMs 2012-01, 2013-01, 2014-01, 2015-01, 2016-01, 2017-01)	3-year measure (2010-2012) aimed at reducing bigeye mortality by 30% and ensuring no increase in yellowfin mortality relative to 2001-2004. Spatial components: • FAD closure for PS fishery within PNA EEZs, 20ºN-20ºS (1 July – 30 Sept)[1] • Closure of two high seas pockets bounded by PNA EEZs from January 2010 onward[1] • FAD closure for PS fishery on HS, 20ºN – 20ºS (1 Aug – 30 Sept) • WCPFC member countries to ensure effectiveness of these measures is not undermined (i.e., avoid transferring effort to areas outside 20ºN-20ºS)	• Strong reduction in bigeye catch during FAD closure periods (had the greatest impact on addressing overfishing relative to other measures) • Overall increase in the size of bigeye landed since more fishing occurred on unassociated schools (i.e., less juvenile catch), which potentially offsets revenue lost through declines in total catch volume • Impacts of closure more moderate for skipjack and yellowfin (i.e., catches not as substantially reduced as for bigeye) • No apparent redistribution of fishing effort from HS pocket closures • Model projections suggest a total PS closure compared to FAD closure would result in only a small additional reduction in bigeye mortality	WCPFC (2008), SFP (2012)

RFMO	Species	Measure	Summary	Results	Source(s)
WCPFC	Bigeye, yellowfin and skipjack tuna	CMM 2017-01	Replaces CMM 2008-01 and all subsequently adopted CMMs (see previous) and extends PS closures, i.e.: • Annual prohibition on PS FAD fishing (July – Sept) between 20°N and 20°S (EEZs and HS) • RFMO members also required to choose between additional 2-month HS FAD closure (April – May or Nov – Dec) • HS pockets bounded by PNA EEZs remain closed	Spatial management combined with other measures have contributed to: • Overfishing of WCPFC bigeye no longer occurring (i.e., F/F_{MSY} = 0.66) • Skipjack and yellowfin stocks also considered healthy (i.e., not overfished and overfishing not occurring)	McKechnie et al. (2016, 2017), Tremblay-Boyer et al. (2017)

[1]These measures were adopted largely as a result of the Parties to the Nauru Agreement (PNA) countries' Third Implementation Arrangement, which delineates specific additional measures for distant water countries fishing access and behavior within PNA waters (see Hanich et al. 2010).

CHAPTER 7
CONCLUSION

With respect to improving the transparency and sustainability of global fisheries Automatic Identification System (AIS) data has been cited as a promising new tool with the potential to expand and transform our current knowledge of global fisheries, opening up new possibilities for monitoring, enforcement and management (McCauley et al. 2016, Kroodsma et al. 2018). The overarching goal of my thesis was therefore to develop and explore the applicability of AIS vessel tracking data to address current problems in marine conservation and fisheries management, specifically (i) interactions of fishing vessels with large-scale MPAs and (ii) the transshipment of fish catch from fishing vessels to cargo vessels at sea. Improved understanding of how fisheries and marine spatial protection influence each other will not only help to better manage both but might also contribute to greater effectiveness of MPAs and potential benefits for surrounding fisheries. Global as well as regional analyses of fishing vessel movements and behavior can help to highlight fisheries patterns and problems around protected areas. Similarly, improved understanding of the role of transshipment of catch at sea in international seafood supply chains will contribute to increased seafood traceability and furthermore reveal priority fisheries and areas for improved fisheries management and trade regulations.

Vessel monitoring data challenges bad practice

Traditionally, spatial and temporal distribution of fishing effort has been mapped based on information from vessel logbooks, observer, and landings data, later complemented by tracking information through VMS (Gerritsen & Lordan 2011). While this gives an impression of global fishing effort, it is rather imprecise especially on smaller scales, and the data is often difficult to access mainly due to privacy concerns. With the use of AIS data for research and marine conservation, an open-access, near real-time tool became available to scientists and the public. While early work mostly used AIS data for vessel localization (Cairns 2005, Cervera & Ginesi 2008), the analysis and classification of different vessel behaviors has become a focus of recent AIS-based research (Natale et al. 2015, de Souza et al. 2016, McCauley et al. 2016, Le Guyader et al. 2017, Ferrà et al. 2018, James et al. 2018).

In this thesis I developed comprehensive approach to analyze AIS vessel tracking data for three different fishing gear types, trawls, longlines, and purse seines (Chapter 2). Using data mining and a Hidden Markov model, I showed that different fishing vessel types can be identified, and their activity classified on a global scale based on their movement patterns. This work was amongst the first attempts to analyze this kind of data in an ecological context and set the foundation for further method development and applications, ultimately supporting neural network-based analyses now applied at a global scale by Global Fishing Watch (www.globalfishingwatch.org).

Large marine protected areas can benefit fish and fisheries

AIS data is now being used to study interactions between global fisheries and marine conservation measures such as MPAs, from individual areas to a global scale (McCauley et al. 2016, Boerder et al. 2017, White et al. 2017). The movements and behavior of fishing vessels can not only inform about fishing fleet dynamics, but may also serve as a proxy for the spatial distribution of fish stocks, especially when combined with catch data (Stelzenmüller et al. 2008).

Examining tuna fisheries around one of the world's oldest LSMPA, the Galápagos Marine Reserve (Chapter 3), founded in 1998, I concluded that the spatial and temporal patterns of fishing effort indicated effects of the marine reserve on tuna stocks in the immediate region. Observing an attraction of purse seine fishing effort to the reserve boundaries (vessels fished four times more within the first 20 km from the boundary compared to an area extending 400 km from the boundary) and linking AIS data to long-term, on-board observer and catch data, it appeared that the Galápagos Marine Reserve positively influenced tuna stocks and associated fisheries in the area: Traditional fishing hotspots had shifted closer to the reserve after its establishment and catches in the vicinity were higher and more stable compared to overall declining catch trends in the wider Eastern Tropical Pacific. Protecting a presumed yellowfin tuna spawning ground (Kliffen & Berkes 2015) as well as unique coastal and oceanographic habitats, the Galápagos Marine Reserve seems to benefit biodiversity conservation and tuna fisheries alike.

However, not all LSMPAs resemble the Galápagos Marine Reserve, especially given their much younger age on average (36 out of 39 LSMPAs were promised or designated after 2008). Extending the analysis to thirteen LSMPAs around the globe (Chapter 4), I found that the landscapes of fishing around each LSMPA were dominated by a unique combination of local factors, but that maritime zoning, mainly the distinction between national waters and the international waters of the high seas, drove regional fishing patterns around seven out of thirteen LSMPAs. This demonstrates not only the importance of maritime zoning, but also the feasibility and impact of large-scale regulations of fisheries. In the context of LSMPAs, the presumed difficulty of monitoring and management of fishing and other activities over an area so large has been cited as an argument against large-scale spatial protection (De Santo 2013, Wilhelm et al. 2014). Automatic Identification System (AIS) and Vessel Monitoring System (VMS) data, as well as other long-range observation and remote sensing tools such as Visible Infrared Imaging Radiometer Suite (VIIRS), a scanning radiometer able to detect visible and infrared imagery, and Synthetic-Aperture Radar (SAR), a satellite-based radar capable of creating three-dimensional images, can bridge this gap from space (Elvidge et al. 2018, Kanjir et al. 2018). The usefulness of AIS has been shown e.g. to monitor effects of the closure of the Phoenix Islands Protected Area in 2015 (McCauley et al. 2016) and has featured in the chase, detention, and eventual conviction of illegal fishing vessels (e.g. the case of the *Marshalls 203;* Pala 2018).

Vessel tracking data and knowledge on the spatial distribution of fishing effort can also contribute to planning and decisions on the siting of future MPAs. It has been remarked that some LSMPAs have been preferably created in lightly used areas where it was easiest to designate them, and not where protection was most needed or useful (Agardy et al. 2011, Craigie et al. 2014, Devillers et al. 2014), a topic heavily debated in the scientific community (McCauley et al. 2014, Singleton & Roberts 2014, MPA News 2018, O'Leary et al. 2018, Rocha 2018). While the motivations for creation certainly vary between LSMPAs, the fact that remote areas with the least impact on stakeholders, the "low-hanging fruit" of marine conservation, tend to be protected first does not render established LSMPAs useless. Established and well-enforced LSMPAs in remote areas can be seen as "pro-active" protection of largely intact areas, which, in the light of rising demands for seafood, are likely to be exploited in the future (McCauley et al. 2013, O'Leary et al. 2018). However, ideally, this should be balanced by protection of areas experiencing intense extractive uses and complemented by adequate fisheries management on a shorter time scale (Toonen et al. 2013, Davies et al. 2017). Knowledge about the distribution of fishing effort from AIS and other tools can help to determine usage of an area and put it into context (Chapter 4). Using AIS data I found that fishing intensity in and around thirteen LSMPAs across the globe varied by a factor of 10, indicating that while some LSMPAs might be placed in low-usage areas, this is not true for all. Extending this to 26 LSMPAs designated by 2018 for which data was available, fishing intensity in a 500 km radius from the boundaries varied by a factor of 200, highlighting the difference between areas such as the remote Antarctic Ross Sea, the world's largest MPA, to the heavily-fished region of the Phoenix Islands Protected Area. This indicates that fishing intensity alone cannot, and should not, be the sole indicator in decisions where to place protection, as the Ross Sea might currently experience relatively low exploitation but is undeniably a fragile and valuable area to protect (Howard 2016). Vessel monitoring data can contribute to these decision processes best when combined with other environmental data layers, to form a wider picture of pressures and threats to inform on priority areas for future conservation.

In this context, two further findings from my work presented in Chapter 4 are relevant: Fishing effort appears to be attracted to older LSMPAs, a result corroborated by previous studies (Edgar et al. 2014, Friedlander et al. 2017), and to those established by low-income countries. In the light of a growing demand for seafood, the potential fisheries benefits of mature MPAs become ever more important, as does their monitoring and enforcement, especially for less wealthy nations which are often highly dependent on their marine resources. Tools such as AIS enhance the capabilities of these nations to implement monitoring and control in their waters and for their MPAs and can aid with establishment of LSMPAs even in areas beyond national jurisdiction (Dunn et al. 2018).

Many of the marine resources on which nations such as Small Island Developing States rely, are highly migratory species such as tuna, billfishes, and sharks. While the usefulness of MPAs for coastal species has been well established, benefits of spatial protection such as offshore LSMPAs for pelagics, especially highly migratory fish species, is unclear due to their extensive horizontal movement patterns (Game et al. 2009). Reviewing the literature for effects of spatial protection measures on large pelagic fishes (Chapter 6), I found that field evidence is scarce, partly due to a scarcity of spatial protection measures for large pelagics, and partly due to the inherent difficulties involved in studying effects of static spatial protection on species roaming thousands of kilometers. However, examining the influence of several species biology and life history traits, it became evident that spatial protection measures can be adapted for many highly migratory pelagic predators. Especially species with predictable migration routes, aggregation sites, as well as limited ranges or homing behavior to specific areas, might benefit more from spatial protection. Tailoring spatial closures to these traits in the form of static and dynamic protected areas, and additionally adapting fisheries management accordingly, could contribute to stock rebuilding as well as enhance fish catches. In this context, vessel monitoring data will be a crucial tool.

Transshipment impairs transparency

Vessel tracking data can be used in many contexts that go beyond observing the actual fishing activity. Transshipment, the offloading of fish catches from fishing vessels to refrigerated cargo vessels, is an important step in global seafood supply chains and links fishing to international markets, yet it is poorly monitored, studied, and regulated (Ewell et al. 2017). A widespread and common practice worldwide, transshipment can facilitate the mixing and subsequent laundering of illegal with legal fish catch, impairing traceability of catches. Using AIS data to document encounters between fishing vessels and cargo ships in the context of seafood supply chain transparency, I highlighted potential shortcomings of current regulatory systems and their implications especially for high seas fisheries (Chapter 5). The fact that transshipment activities concentrate in a few global hot spots, such as the Russian EEZ, off West Africa, and in the Tropical Pacific, and primarily involve trawlers and longliners, showed that transshipment is more important in some fisheries and regions than others. It furthermore demonstrated that regulations, such as the prohibition for purse seiners to transship at sea in large areas of the ocean (WCPFC 2009, Ewell et al. 2017), can be effective, as transshipment from purse seiners was only observed in a few cases. However, mismatches in the comparison of AIS observations of transshipment events with official documentation through RFMOs and industry also highlighted the value of a combination of multiple documentation systems, especially independent ones such as AIS.

Where to go from here

This work is one of the first in the emerging field of applying vessel tracking data in the context of marine conservation and many questions yet remain to be addressed by using AIS data and other promising long-range observation and remote sensing tools. I am confident that the advent of these tools is just the start of a new era in marine science and spatial management. However, a range of problems still need to be addressed:

A big gap in current AIS data and subsequently research is the lacking coverage of small-scale, inshore fishing fleets, which make up the vast majority (86% of motorized vessels are shorter than 12 m overall length) of fishing fleets worldwide (FAO 2018). The fact that more than 99% of all MPAs (by number; UNEP-WCMC and IUCN (2018) Marine Protected Planet and O'Leary et al. 2018) are located in coastal waters, highlights the need for improved monitoring of these small-scale fleets. However, often vital information on the highly diverse small-scale fleets such as number of fishermen and boats, fishing types, and catches, are missing or incomplete (Chuenpagdee et al. 2006). This lack of knowledge not only often results in a marginalization of small-scale fishing fleets regarding management planning and decisions (Chuenpagdee et al. 2017), but also leads to their underrepresentation in catch statistics as well as impact analyses (Pauly & Zeller 2016, Selgrath et al. 2018). Surveillance, and the lack thereof, has been identified as one of the key challenges in small-scale fisheries management and governance (Salas et al. 2007). Yet, next to their sheer numbers, the lack of a constant power source on most of small fishing boats renders monitoring tools such as AIS and VMS difficult to implement. Research and development is underway to deploy autonomously powered vessel tracking tools for these small vessels (James et al. 2018) and integrate it with existing monitoring, management, and marine spatial planning. In a next step, a planned online platform specifically designed to facilitate management of MPAs will take vessel monitoring data to the next level. This platform will combine novel and existing small and large-scale vessel tracking data and additional environmental and socio-political information and operationalize it for marine spatial planning such as static and dynamic MPAs, fisheries closures, and fisheries management. It will also allow to repeat analyses trialed in this work in the future in more detail and with additional and better data to get a broader, long-term understanding of interactions of fishing fleets with marine conservation measures and fisheries management. An example of additional data improving current observations is the VIIRS data, detecting lights of vessels at night to inform on the 'dark fleet', vessels not carrying AIS and subsequently not appearing in the AIS data (Fig. 7.1). Data like these can be useful to monitor use of AIS especially for smaller vessels, complete fishing activity maps, increase observance of vessel presence and absence in areas of interest such as fishery closures (Elvidge et al. 2018), and get a better understanding of gaps of AIS coverage. It also increases understanding of the behavior of specific fishing fleets such as squid jiggers (Miller et al. 2013), which use bright lights to attract squid.

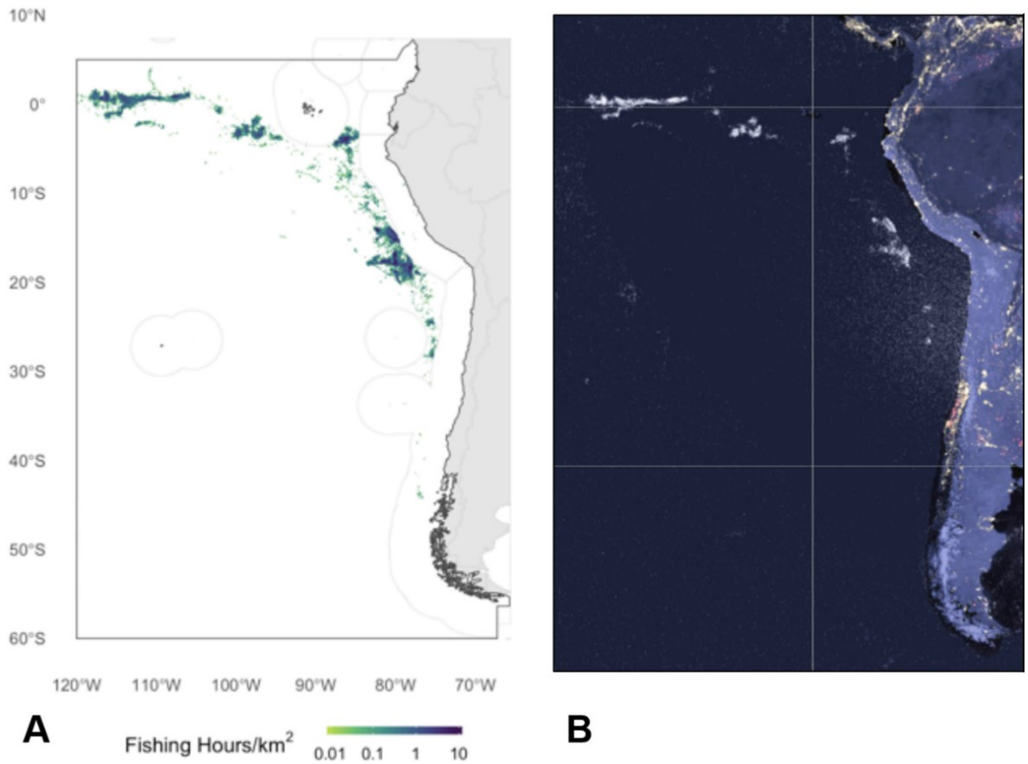

Fig. 7.1 Comparison of AIS data (A) with VIIRS data (B) off the west coast of South America for the year 2017. Note the detection of vessels especially off Chile as well as further offshore by VIIRS not showing up in the AIS data. AIS data by Global Fishing Watch, VIIRS data by NOAA.

In the light of expanding fisheries, increasing number and coverage of MPAs, as well as the negotiations for an international high seas treaty to protect biodiversity in areas beyond national jurisdiction (United Nations General Assembly Resolution 69/292), the importance of remote sensing and monitoring tools such as AIS to support monitoring, control and surveillance efforts on national and international levels will only increase with time (Dunn et al. 2018). Applications range from the monitoring of fisheries in national waters and the high seas, assessment of fleet behavior around conservation measures such as MPAs and fisheries closures, and observation of transshipment events at sea, to combating illicit activities such as illegal fishing, smuggling, forced labor, and trafficking of humans, drugs, and weapons (International Labour Office 2013, Longépé et al. 2018). Further applications include the monitoring of other vessel types than fishing vessels to analyze effects of shipping on marine mammals (e.g. ship strikes, Conn & Silber 2013), map underwater noise and pollution risks (e.g. oil spills, discharge of ballast water in the context of introduction of invasive species), observe marine tourism and other marine industries such as oil and gas exploration, drilling, and mining, as well as aid with search and rescue missions (Carson-Jackson 2012).

However, for AIS to reach its' full potential for monitoring and enforcement, a number of regulatory changes and policy interventions are required, as the tool is only as good as the legislation supporting it – while the International Maritime Organization (IMO) mandated AIS use for fishing vessels larger than 300 gross tons undertaking international voyages, it's use and actions in case of mis-use, tampering, or lacking compliance are little specified and regulated. In addition, legislation for AIS use in national water vary strongly by country: While, for example, in the European Union all fishing vessels longer than 15 m in overall length are required to carry and operate AIS (EU Dir 2011/15/EU), Canada has exempted all fishing vessels moving within national waters from carrying AIS (http://www.ccg-gcc.gc.ca/eng/CCG/Maritime-Security/AIS#guidelines). Thus, a large number of vessels not carrying AIS still go "unseen", creating gaps and blind spots in analyses (McCauley et al. 2016, Kroodsma et al. 2018). Furthermore, AIS information should be linked with fleet registries to enable matching of AIS signals to vessel identities – to date, unlike unique and permanent IMO numbers and call signs, AIS maritime mobile service identity (MMSI) numbers, the numbers unique to every AIS transponder, can be changed and are not recorded on a comprehensive, global scale (Dunn et al. 2018).

If these conditions are improved, vessel tracking and monitoring, as shown in this and further work, have the power to change the face of marine spatial management, and to contribute to a new era of improved marine governance. To borrow a quote from SkyTruth, one of the founding organizations of Global Fishing Watch:

"*If you can see it, you can change it.*"

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Appendix E

Several other works I have co-authored are relevant for, but not included in this thesis. They have been submitted as electronic supplements.

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